CAN WE AFFORD TO DELAY ACTION AGAINST WEEDS IN VALUED NATURAL AREAS?

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Abstract The use of thresholds in the management of agricultural weeds has been under consideration for some years. In a recent paper Panetta and James (1999) put forward an argument that, for a number of reasons, an action threshold based upon the actual, realised impacts of a weed within a natural ecosystem is not practicable. Alternatively, it appears that actions for weed control should be triggered at low densities of the most serious environmental weeds. This is because it is likely that the cost-effectiveness of weed control efforts is maximal during the earliest stages of weed invasion. The present paper explores this hypothesis, using information on treatment costs and effectiveness.

INTRODUCTION

Weed invasions pose a serious threat to the values of natural ecosystems (Humphries *et al.* 1991). Such invasions are considered second only to habitat destruction in terms of overall threats to global biodiversity (Walker and Steffen 1997). Weed invasions have been linked with major changes in the structure and composition of natural ecosystems, as well as with disruption of key ecosystem functions (see Panetta and James (1999) for references).

A key consideration in managing weed invasions of natural ecosystems lies in deciding at what point intervention should occur and to what extent. No doubt the most cost-effective option is to prevent invasive species from entering the country altogether (Hobbs and Humphries 1995), but this does not assist land managers in their attempts to deal with widespread, serious weeds. In order to rationalise the use of scarce resources, a priority must be placed on managing environmental weeds where they threaten areas of high value; this paper relates specifically to weed management in such areas.

Henry (1994) appears to have been the first to suggest that the management of weeds of natural ecosystems could be approached more effectively by attempting to keep weed populations below a threshold that would cause “native plant loss or other ecosystem degradation”. This approach would be facilitated by a shift in management research to determining the requisite thresholds (Henry 1994). More recently, Adair and Groves (1998) maintained that threshold levels for declines in biodiversity could be used as a basis for setting the “maximum tolerable level of infestations for nature conservation purposes”. However, Panetta and James (1999) have argued that any serious attempt to define action thresholds on the basis of weed impact must take into account: a) the benefits provided by the system being managed; b) damage relationships resulting from the presence of weeds in the system; c) population dynamics of the weed(s) concerned; and 4) the treatment of risk. Since this information is not readily available (or easily obtained) for any weed in any natural ecosystem, they recommended the use of an action threshold (Cousens 1987) that could be identified at a relatively early stage of the invasion process.

Both the costs and the effectiveness of management efforts targeting invasive species could be expected to vary according to the stage of invasion at which intervention occurs. Management effectiveness must be considered over the longer term, generally in relation to the return of the natural ecosystem to some semblance of its original composition, structure and functions performed. The aim of this paper is to examine the costs associated with managing weed infestations, and to appraise the potential effectiveness of intervention at different stages of weed invasion.

HOW REVERSIBLE IS WEED IMPACT?

A number of papers have highlighted the importance of managing sites such that the weeds removed are replaced by desirable species (e.g. Groves 1990, Panetta and Lane 1996). Groves (1990) argued that a better understanding of the biology of potentially competing indigenous species would provide a basis for promoting species that could usurp the resources freed by weed control, so preventing either a reinvasion by a targeted species or invasion by another. Ideally, replacement of the weeds would occur by spontaneous regeneration that occurred via propagules of native species, either freshly produced, stored on plants or within a soil seed bank. However, where weed infestations are extensive and/or longstanding, the source(s) of viable propagules may be highly deficient or lacking altogether. Such sites may have passed what Aronson *et al.* (1993) have termed a "threshold of irreversibility". Adair and Groves (1998) suggest that this situation might occur where weeds have dominated vegetation types characterised by obligate re-seeders with short-lived seed banks and where seed immigration from neighbouring areas is unlikely.

The approach to an irreversible (or reversible only with considerable difficulty and cost) situation has been shown by a study of the effects of the invasive species *Acacia saligna* (Labill.) Wendl. upon fynbos communities in the Cape Peninsula, South Africa (Holmes and Cowling 1997a,b). In this study, two measures of diversity (species richness and Shannon Wiener index) of the seed bank decreased markedly as the infestation developed (Table 1). It is important to note, as well, that in the course of the invasion large, persistent seed banks of the weed had developed, so that any germinants of native species following removal of the weed would be likely to be swamped by massive germination of the highly competitive *A. saligna*.

Several Australian studies have also demonstrated the relative lack of representation by native species in the seed banks of sites long-dominated by invasive plants. Panetta (1982) documented the domination of seed banks by exotic species in Victorian sites infested by *Rubus polyanthemos* Lindeb. In the Northern Territory Lane *et al.* (1997) noted that while there were no measurable effects of young stands of *Mimosa pigra* L. (2-3 years old) on the composition, general abundance and patterns of emergence of native herbaceous species present in the soil seed bank of floodplain vegetation, this weed had depressed the richness and density of soil seed banks under 10 year-old stands in the same region (unpublished, quoted in Lane *et al.* (1997)). Thus the depletion of seed banks of native herbaceous species by *M. pigra* was a damage function that operated over a number of years.

Where seed banks are highly depauperate and there is no local source of propagules, options for treating infested sites include the sowing of seed and planting of seedlings of desirable native species. Panetta and Groves (1990) have addressed a number of factors to be considered when developing revegetation strategies following weed control. Suffice it to say here that reintroducing species by seed, while less costly than reintroducing species by planting seedlings or more advanced growth, is generally a riskier strategy, with regeneration failures common, particularly in areas with unpredictable or erratic rainfall regimes. Furthermore, however advanced the growth of native species upon their (re)introduction, additional costs are incurred, over the short term at least, by the necessity to reduce the impact of residual weed growth upon the establishment of desirable species (Panetta and Groves 1990).

Table 1. Diversity measures for seed banks in fynbos communities uninvaded, recently invaded and long-invaded by *Acacia saligna* (from Holmes and Cowling 1997a). Recently invaded sites had been infested for between 1-2 fire cycles (≤ 25 yr) and long-invaded sites by more than 2 fire cycles (> 25 yr).

Site no.	Uninvaded	Recently invaded	Long-invaded
1			
S[a]	10.5	5.18	3.32
H'[b]	**1.96**	**1.22**	**0.92**
2			
S	8.90	7.85	2.45
H'	**1.88**	**1.80**	**0.63**
3			
S	10.5	7.12	5.10
H'	**1.80**	**1.50**	**0.90**

[a] Number of species per unit area (excluding *Acacia saligna*).
[b] Shannon Wiener index.

MANAGEMENT COSTS

From the discussion above, it is obvious that where weed invasions have been allowed to proceed to a significant extent, the total cost of intervention will comprise not only the costs of treating the weed(s), but will also involve costs of revegetation and its subsequent maintenance.

Unfortunately, there are few readily available data on the costs of treating weed infestations, and such data often relate to vastly different spatial scales. For this discussion I will present examples from both a coarse (single site) and fine (continental) scale. In terms of the former, the relationship between infestation density and cost of treatment has been shown to depend upon the type of treatment involved. In the KwaZulu-Natal region of South Africa, for example, there were substantial increases in the costs of treating tree weeds with increasing weed density, although the labour required for application of chemical to cut stumps was not nearly as responsive to density as was a tree felling operation (Fig. 1). Further contributions to the cost of treating sites in this region included variables that would impede progress, such as slope, type of terrain and access time – the percentage labour-day time travelling between a base and the treated site (Goodall and Naudé 1998). This last variable may be a major cost component for remote sites.

A key finding of this South African work was the identification of a "maintenance" weed density class, comprising between 0-5% cover, where a low annual or biennial control commitment (depending upon the population dynamics of the targeted species) would be sufficient to prevent substantial population increase. Presumably, the impact of the weed upon ecosystem composition, structure and function would be minimal at such low densities, as well. This concept of "maintenance densities" was adopted by Panetta and James (1999) in their argument concerning the stage(s) of invasion when intervention might be best timed. It is also consistent with the view expressed by Hiebert (1997) that the urgency of weed control (defined in terms of the amount of increase in effort required to achieve successful control following a delay in action) is an important factor in prioritising weed control efforts.

In the United States, Wilcove and Chen (1998) have recently calculated the cost of maintaining federally listed species threatened by non-indigenous species (NIS) and fire. These costs amounted to $US32-42 M

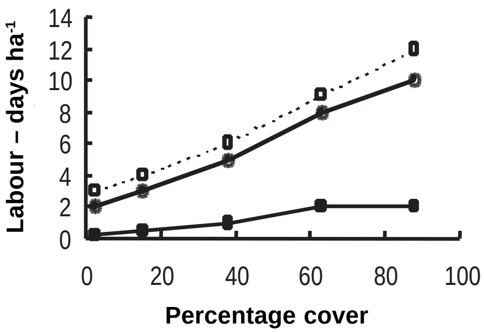

Figure 1. Average work rates for normal conditions for control actions in relation to infestation density of tree weeds in study sites in KwaZulu-Natal (from Goodall and Naudé 1998). Control actions are: □ fell only; ● stem injection; and ■ application of herbicides to cut stumps.

per annum. In relation to NIS, it was considered that management costs for some endangered species would decline over time once currently neglected problems were brought under control. For example, in North American sites where tamarisk (*Tamarix* spp.) has not been controlled for years, the cost of removal can be as much as $US1700 ha^{-1} in the first year, decreasing to below $25 ha^{-1} in the second year. Subsequent maintenance requires an expenditure of less than $25 ha^{-1} every 2-3 years (Wilcove and Chen 1998). These follow-up and maintenance costs are for spot herbicide treatment. Should control be neglected following the initial 2 years of treatment, it is believed that within 10-15 years, the original treatment of manual cutting and removal would be required again (C. Deuser, unpublished data, cited in Wilcove and Chen 1998). For a variety of weeds and natural areas in North America, L. Chen (unpublished data) has shown that initial control costs were 1.8 to 350 times greater than the associated maintenance costs, suggesting that early intervention would represent a significant investment. To such monetary costs must be added other, non-monetary costs of losing endangered species; Wilcove and Chen (1998) have highlighted the potential for local extinction of endangered species if weed infestations are allowed to develop unchecked.

In both of the examples given, it would appear that early intervention was the most economical and cost-effective option. With such timely intervention, not only are the direct costs of weed control lowered, but the necessity to incur costs through revegetation efforts should be minimised, if not avoided entirely. This conclusion is supported by recent American work (Smith *et al.* 1999) that showed that "an early and vigorous approach to the eradication of new invasive weed infestations is expedient, for both environmental and economic reasons". While I do not wish to argue that eradication is generally the best approach to managing new invaders, the point has been made elsewhere (Panetta and James 1999) that the difference between the effort and resources expended in an unsuccessful eradication effort and those expended in a successful attempt to confine a weed to the maintenance level (Goodall and Naudé 1998) may not be particularly great.

Higgins *et al.* (in press) have recently explored a range of strategies and funding schedules for clearing alien plants in fynbos communites of the Cape Peninsula, South Africa. They found that clearing strategies that prioritised low-density sites dominated by juvenile alien plants proved to be the most cost-effective. Delaying the initiation of clearing operations had the strongest effect on both the eventual costs of the clearing operation and the overall threat to native plant diversity.

BENEFITS FROM DELAYING ACTION

Three potential benefits from delaying action can be identified. Firstly, if nothing *at all* is known about the potential impact of a newly invading weed species, a failure to intervene may allow for observations and assessment of such impact. This approach may have value in some situations, given that the majority of naturalising species is considered to have only minor ecological effects (Williamson 1996). It should not be employed, however, where weeds that are known to have serious impacts are concerned (Panetta and James 1999).

Secondly, delaying action will generate scope for allocating resources to alternative projects. This relates to the so-called "opportunity" costs associated with weed management actions. The importance of this factor will be determined by the alternative demands for funding, as well as the combined values of the particular site to be treated.

The last area of potential benefit subsumes the savings that could be achieved by discounting, should action be delayed. The size of the incentive for delaying intervention depends upon the discounted cost of the delay. Evaluating whether it makes sense to delay depends on how rapidly the costs escalate due to delaying, as well as the discount rate. Results from Higgins *et al.* (in press) suggest that policy makers who use discount rates greater than 0.04 would have a limited perception of the cost of delaying clearing operations. Lower discount rates (ca. 0.03) are usually recommended for environmental projects; for these the financial incentive for delaying is weak (see Pearce and Turner 1990, Pearce 1993).

Whatever discount rate may be best applicable, Higgins *et al.* (in press) stated that the cost and uncertain success of projects that aim to restore locally extinct populations further calls into question the wisdom of delaying intervention. With regard to the reintroduction of rare grassland species in western Victoria, Morgan (1999) observed that the failure of most reintroduced populations to recruit threatened the long-term persistence of reintroduced plantings. He suggested that the conservation and management of remnant populations *in situ* is critical to the survival of rare species in natural habitats. We cannot assume that we can readily reverse the local extinctions that may result from weed invasions!

MANAGEMENT FOCUS

In his landmark letter, Henry (1994) maintained that the alternatives in managing weeds of natural areas were either to keep them free by "…traditional labor-intensive gardening/agricultural practices, making these areas really gardens, parks, preserves, etc.", or to keep weeds "…as a population below a threshold that will cause native plant loss or other ecosystem degradation." Panetta and James (1999) have argued that such thresholds are difficult, if not impossible, to identify and that action to manage serious weeds should occur well before the point at which they are causing measurable impacts. The need to target these species means that, with the exception of instances where management tools such as fire and biological control confer adequate suppression, land managers are forced to utilise labour-intensive techniques.

Henry's preference for avoiding labour-intensive practices can be related to a recurring theme in recent writings on the management of invasive species: that ecosystems should be managed in such a way as to minimise their invasibility, in particular through managing various forms of disturbance, rather than focussing upon individual invaders (see Hobbs and Humphries 1995). While this approach has validity, it is over-idealised, in that many invasive species may succeed under disturbance regimes that are designed to favour indigenous plants, and an ever-expanding suite of invasive species is bound to contain plants that are adapted to any of a variety of disturbance regimes. For these reasons, Panetta and Lane (1996) argued that it is not appropriate to force a choice between an "ecosystem focus" and an "invader focus"; more effective management of invasive plants should arise from a combination of activities that aim to reduce community invasibility, coupled with control actions targeting priority invasive species.

CONCLUSIONS

It does not appear practicable, or indeed desirable, to utilise action thresholds based upon actual, realised weed impact. Rather, the most cost-effective approach to the management of serious weeds will likely incorporate actions that are triggered at relatively low weed densities. The effectiveness of this approach arises from a minimisation of weed impacts, both upon the ecosystem in general, and upon threatened species in particular. Savings that may arise as a result of not deferring management intervention to more advanced stages of weed invasion should further contribute to cost-effectiveness.

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