Weed control thresholds: a useful concept in natural ecosystems?

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Summary

The threshold concept has received considerable attention in relation to the management of well-established weeds and other pests in agricultural systems, but its applicability to the management of weeds of natural ecosystems has yet to be examined in any depth. Four aspects that must be considered in relation to the use of thresholds in weed management are identified, namely:

i. **benefits provided by the system being managed,**
ii. **damage relationships resulting from the presence of weeds in the system,**
iii. **population dynamics of the weeds concerned, and**
iv. **the treatment of risk.**

It is argued that difficulties associated with capturing much of this information would generally preclude such an approach to determining action thresholds for environmental weeds. Significantly, the debate continues as to the usefulness of the threshold concept in the context of weed management in agricultural systems. Recent Australian research has shown that the greatest economic benefits in cereal crops are provided by management actions that aim to minimize weed seed bank populations.

Weed management in natural ecosystems is often a very labour-intensive undertaking and the costs of weed control per unit area can escalate rapidly with increases in weed density. Control efforts are arguably most cost-effective during the earliest stages of weed invasion; management effort should be focused here, given that particular weeds are considered sufficiently serious to warrant intervention (and the natural area considered sufficiently important to warrant protection). An understanding of weed population dynamics may allow the definition of maintenance levels, where a low annual or biennial control commitment would be sufficient to prevent substantial population increase. Better information on the cost (and effectiveness) of control efforts at different stages of weed invasions should assist in defining such levels, which could function as triggers for weed control.

Introduction

Weed invasions pose a serious threat to the values of natural ecosystems (Humphries *et al.* 1991). Invasions by these so-called 'environmental weeds' have been linked with major changes in the structure and composition of natural ecosystems, as well as with disruption of key ecosystem functions (Vitousek 1986, Humphries 1993, Cronk and Fuller 1995, Walker and Smith 1997). In contrast to agricultural ecosystems, where there are targeted funding, programs and corporate interests dedicated to weed management, the funding and programs devoted to the management of environmental weeds are meagre (AACM International 1997). For example, 1995/6 expenditure on environmental weed control in protected areas was $0.60 and $0.38 per hectare in Victoria and New South Wales respectively (ENRC 1998, Leys 1996 and personal communication). The scarcity of resources available for managing this threat to natural ecosystems highlights the requirement for a well-considered approach to the problem. Such an approach must address a range of needs, from the development of a procedure to assist in identifying and ranking weed problems of potential national significance (Panetta and Lane 1996, Virtue *et al.* 1998), to the smaller-scale considerations of determining which sites warrant intervention (Shaw 1994, Goodall and Naudé 1998), prioritizing weed species for management on a site-by-site basis (Williams 1997) and evaluating the feasibility of obtaining positive management results (Hiebert 1997).

Weed management programs can be divided into three types (Cousens 1987): eradication (where the objective is to totally eliminate the pest, avoiding the necessity of future control efforts), prophylaxis (where the objective is to minimize the risk of damage through the regular use of control measures, regardless of the size of the pest population) and containment (where the intention is to keep the pest population at or below a specific level or within a particular area). The last approach involves the acceptance of some level of impact or damage, utilizing intervention only where this is justified. A containment approach is based on the assumption that a weed population that justifies intervention (i.e. a threshold) can be defined and that weed populations will be monitored as a basis for decision making (Cousens 1987).

The threshold concept has traditionally been employed in decision-making in the context of pest management in agricultural production systems (Norton and Mumford 1993). As with many terms, a threshold can be defined in a number of ways, leading to substantial confusion in the literature. Cousens (1987) argued that since the purpose of a threshold is for use as a guide in determining when action to control a pest is to be taken, the relevant generic term should be action threshold. This too can be identified in a number of ways (Cousens 1987).

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, where serious weeds are involved, the most cost-effective option is to prevent invasive species from entering the country altogether (Hobbs and Humphries 1995). While procedures have been put in place in Australia to minimize the active importation of invasive species (e.g. Walton and Ellis 1997), some potentially invasive species are bound to evade quarantine (pre-entry screening) efforts. Of more immediate importance, however, are the large numbers of invasive species already widely naturalized or newly naturalizing in Australia (Humphries *et al.* 1991, Csurhes and Edwards 1998, Groves 1998). It is clear that a priority must be placed on managing environmental weeds where they threaten areas of high value (Shaw 1994, Goodall and Naudé 1998); this paper relates specifically to weed management in such areas.

Henry (1994) suggested 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) have maintained that the determination of threshold levels for declines in biodiversity could be used as a basis for setting the 'maximum tolerable level of infestations for nature conservation purposes', particularly for serious weeds that are very widespread. Furthermore, they argued that reducing weed infestations to well below such threshold levels is likely to represent a poor use of resources (Adair and Groves 1998).

There is no doubt that thresholds hold an intuitive appeal, but over time there have been doubts expressed as to their usefulness in managing weeds in crops (Cousens 1987, Wallinga and van Oijen

1997), and it is by no means clear whether action thresholds can be readily defined for weeds of natural ecosystems. From our perusal of the relevant literature (e.g. Cousens 1987, Doyle 1991, Norton and Mumford 1993), we conclude that there are four aspects that must be considered in relation to the use of thresholds in weed management. These are:

i. the benefits provided by the system being managed,
ii. damage relationships resulting from the presence of weeds in the system,
iii. population dynamics of the weeds concerned, and
iv. the treatment of risk.

In this paper we will consider these aspects with regard to the management of weeds in both agricultural and natural ecosystems. Our aims are to explore the scope for generalization of the threshold concept from agricultural production to natural areas, and to provide suggestions for the types of information that might be utilized to determine action thresholds for environmental weeds.

Table 1. Representative values of natural areas (modified from James 1991).

Direct uses	Indirect uses	
	Functions and services	Attributes
Timber production	Waste treatment	Biodiversity
Forage production	Nutrient cycling	Aesthetic value
Mineral production	Erosion control	Spiritual value
Wildlife production	Water regulation	Educational value
Fish production	Soil formation	Scientific value
Energy production	Climate regulation	Historic value
Wildflower production	Biological pest control	
Recreation / tourism	Disturbance regulation	
Water supply		

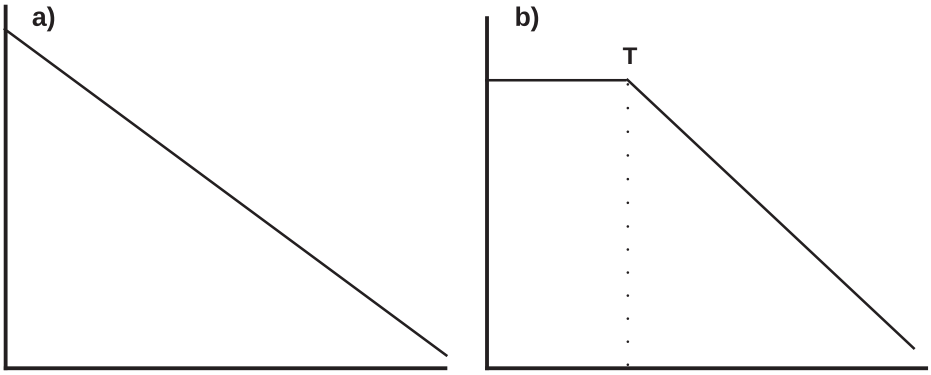

Figure 1. Generalized damage relationships: a) linear and b) threshold (T).

Quantification of benefits

Agricultural enterprises exist to earn returns from the goods produced. The value of agricultural products to the producer is largely determined by market forces and / or subsidies provided by governments. Prices for products may fluctuate considerably over time, but there is no ambiguity as to what the products of agro-ecosystems are. Accordingly, the potential market-based benefits from these systems are relatively easy to define. Returns from investment to reduce the impacts (see below) of pest organisms upon either product quantity or product quality can be calculated (Auld *et al.* 1987). Such calculations are considerably more difficult in grazing systems, owing to a number of problems associated with assigning values to herbage production (Auld *et al.* 1987). This may account for our failure to find any evidence for the use of the threshold concept in pasture weed management.

It should be noted at this point that whereas quantification of the benefits associated with agricultural production can be a relatively straightforward matter, the associated costs (potential groundwater contamination by herbicides used to control weeds, soil erosion resulting from tillage etc.) are often either under-represented or ignored in economic analyses. We will return to this issue later.

Natural ecosystems provide a wide range of benefits, ranging from products (e.g. timber or minerals) that are marketed, to ecological functions and services (e.g. nutrient cycling, water supply and waste treatment) and human values (e.g. cultural and aesthetic). James (1991) makes a distinction between the direct uses of a natural area, involving physical interaction with the area, and indirect uses, which include both functions (services) and attributes (aspects providing for non-material human needs) (Table 1). The values of ecosystem services are mostly outside the market (Costanza *et al.* 1997); values of ecosystem attributes are entirely so.

There has been active development of methods for capturing the values of non-marketed uses of natural ecosystems (e.g. Kopp and Smith 1993, Pearce 1993, Pearce and Moran 1994). Stated preference methods such as contingent valuation, and revealed preference methods such as hedonic pricing and the travel cost method (see Pearce 1993), have all been used to estimate the values of natural areas of various types under various management regimes. However, none has been applied explicitly to value the outcomes of environmental weed control in natural areas. Current Australian research aims to develop a framework and associated measures that can be used to value the impacts of management activities, including control of environmental weeds, in natural areas (R.F. James, unpublished). For the purposes of this paper, however, it is fair to say that there is no widely accepted procedure for obtaining an aggregated value for the array of benefits provided by such areas, or for valuing the consequences of alterations in the levels of those benefits resulting from management activities.

Damage relationships

Weeds in agricultural ecosystems

It is widely assumed that there is a particular level of weed infestation in invaded agro-ecosystems, beyond which weed control activities should be undertaken. Decision-making in relation to pest management in agricultural systems should be based on the form of the damage relationship (Norton and Mumford 1993). Where the relationship between damage and pest density is linear (or approximately linear) (Figure 1a), the slope of the relationship is the critical factor in determining the point at which control should commence and the degree of control required. However, where there is little or no damage at lower pest densities, a biological threshold relationship (Figure 1b) will apply. The latter situation could arise where there is a degree of tolerance to low levels of pest attack, or a crop plant is able to compensate for a certain amount of damage. Where it applies, therefore, the threshold level has a large effect on deciding the required level of pest management (Norton and Mumford 1993).

Suggestions have been made in the literature that the relationship between yield and weed density is sigmoidal (see Cousens (1987) for references), with an identifiable competition threshold. However, Cousens (1985a) demonstrated that in the majority of cases he examined, the relationship between percentage yield loss and weed density was approximately hyperbolic. This relationship has a sound biological basis in that it reflects the fact that the sizes of individual weeds (and thus their competitive abilities) are at a maximum at low density. Weeds thus have an effect on crop yield even at low densities and there is no identifiable threshold damage level. Cousens (1985a) attributed the popularity of the competition threshold to the fact that experiments at low weed density may not show statistically significant differences in yield from a weed-free control, and made the point elsewhere (Cousens 1985b) that in most field experiments even large yield effects may not be statistically significant.

Owing to the difficulty in detecting a biological threshold damage level (Norton and Mumford 1993) in weed-infested crops, an action threshold has been defined, in economic terms, as the weed density (or relative cover – see Berti and Sattin (1996)) at which the cost of control measures equals the increased return on yield in that season that would result from weed control. Control measures become cost-effective at this point, in that they produce an economic benefit (Cousens 1987, Norton and Mumford 1993).

Weeds may be qualitatively different from other pests of agro-ecosystems, in that crops are rarely invaded and significantly damaged by only a single weed species. It is more realistic to consider a weed population as comprising a multi-species assemblage, particularly where dicotyledonous weeds are concerned. Some measure of success has been achieved in modelling the impact of a combination of weed species upon crop production by using relative competitive indices (Coble and Mortensen 1992, Cousens 1992).

Weeds in natural ecosystems

As noted earlier, natural ecosystems provide a range of benefits derived from a variety of direct and indirect uses (Table 1). Damage relationships with respect to these benefits have rarely been explored in natural ecosystems, for a number of reasons, including the following:

- the costs associated with determination of the damage relationships;
- the multi-attribute nature of most natural areas, and hence the existence of multiple damage relationships;
- the potential difficulties in the aggregation of damage relationships in order to determine an overall threshold for a particular weed in a natural area;
- the non-market nature of the benefits which might result from weed control.

Accordingly, it is very difficult at this stage to address the functional forms of such damage relationships, and hence the occurrence or otherwise of a threshold, for natural areas. However, there are some isolated quantitative analyses of the impacts of weeds on some attributes of natural areas. In this section we will present examples relevant to the impacts of weeds upon: a) biodiversity, b) regeneration of native species and c) an ecological service.

Impact of weeds upon biodiversity. In their study of the impact of the increasingly abundant perennial grass *Brachypodium pinnatum* (L.) Beauv. in the species-rich chalklands of western Europe, Bobbink and Willems (1987) determined the relationship between species diversity (as defined by the Shannon index of diversity (H')) and the relative abundance of *Brachypodium* (Figure 2). They argued that once *Brachypodium* reached over 50% of the above-ground biomass, it started to influence the fate of other plant species. This was based upon a strong negative linear relationship ($r^2=0.815$, $P<0.001$) between H' and *Brachypodium* biomass in instances where *Brachypodium* contributed over *c.* 50% of the total above-ground vegetation (Bobbink and Willems 1987). However, this dataset shows no threshold level (cf. Figure 1a); it is clear from the cubic relationship fitted to the entire data set (Figure 2) that the presence of *Brachypodium* was associated with a decline in species diversity even where it occurred at lower relative densities. The choice of '*c.* 50%' as an action threshold was thus rather subjective.

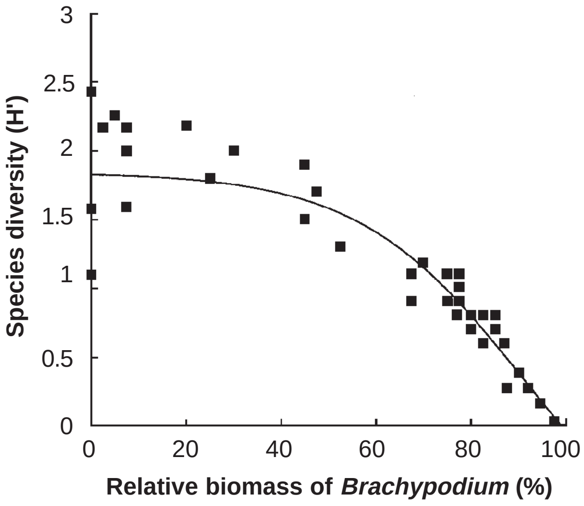

Figure 2. Relationship between species diversity, expressed as the Shannon index (H') and the relative phytomass of *Brachypodium pinnatum* in chalk grassland (redrawn from Bobbink and Willems 1987).

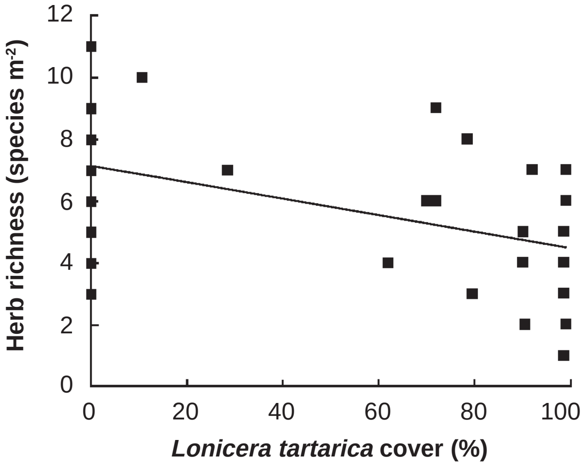

Figure 3. Relationship between herb species richness and *Lonicera tartarica* cover ($r^2=0.25$, $P<0.05$) (redrawn from Woods 1993).

No other data sets comparable to this detailed study on *Brachypodium* appear to be available. Data describing the effects of weeds upon species richness alone, neglecting the relative abundance (or evenness) component, of diversity (Odum 1971) are unlikely to be sufficiently sensitive to be used as a basis for decision-making. For example, decreases in species richness may lag considerably behind decreases in diversity indices in the course of invasion by weeds (Bobbink and Willems 1987). It is also important to note that there has been considerable controversy over the biological relevance of a number of diversity indices, in particular the Shannon index, which was utilized in the *Brachypodium* study (see Hurlbert 1971). Problems of curve-fitting aside, workers must be cognisant of the basis for, and implications of using, particular indices of diversity (Magurran 1988).

Studies investigating the impact of weeds upon species richness have provided no evidence of threshold damage relationships. Woods (1993) reported a statistically significant negative relationship between the cover of the introduced shrub *Lonicera tartarica* L. and the number of native herbaceous species in the understorey of a New England forest (Figure 3), but this relationship failed to explain 75% of the variation in species richness. A similar, but better-defined relationship was found between weed cover and native herbaceous species richness of Western Australian woodlands (Figure 4) but these data in themselves cannot provide a basis for a simple, objective determination of an action threshold.

Impact of weeds upon regeneration of native species. Weeds commonly interfere with the regeneration of native species, affecting both native plant community composition and biomass, thus causing substantial changes in community physical structure and function over time.

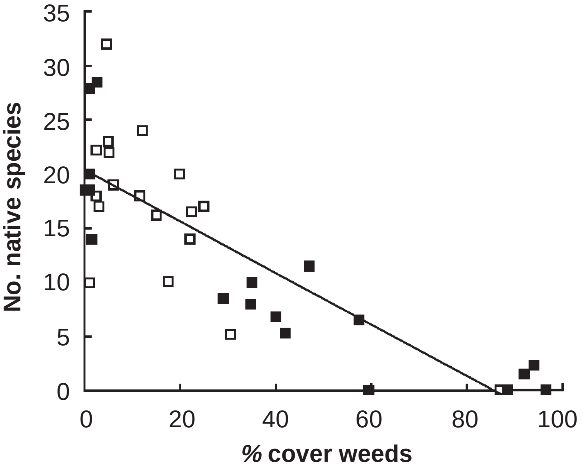

Figure 4. Relationship between species richness of native herbaceous plants and the per cent cover of exotic plants in gimlet *Eucalyptus salubris* F.Muell. (■) and wandoo *E. capillosa* (Brooker & Hopper) (□) woodlands (r^2=0.70, P<0.001) (from Abensperg-Traun *et al.* 1998).

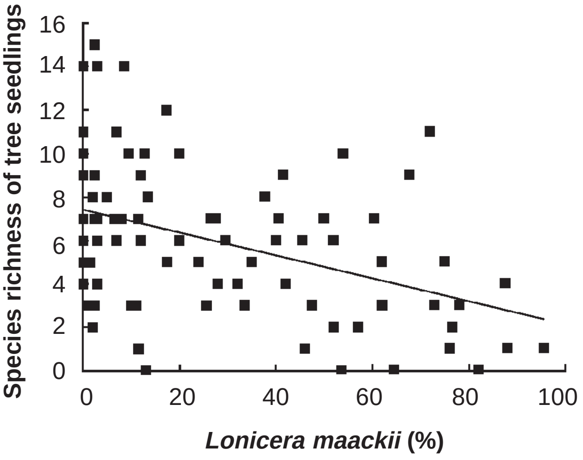

Figure 5. Relationship between species richness of tree seedlings and *Lonicera maackii* cover (r^2=0.152, P<0.001) (redrawn from Hutchinson and Vankat 1997).

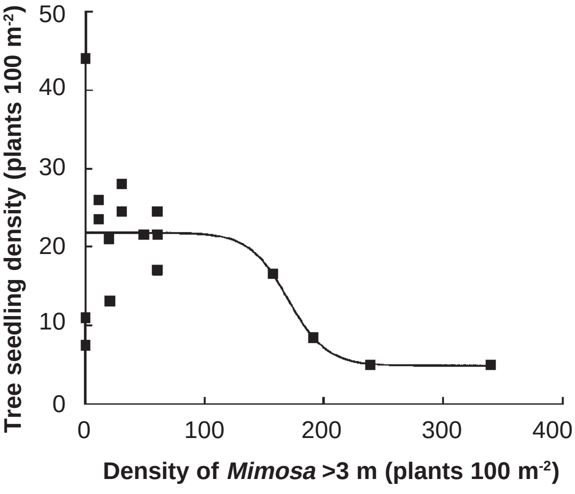

Figure 6. Relationship between native tree seedling density and the density of large individuals of *Mimosa pigra*. Sigmoidal relationship (r^2=0.38, P<0.1) fitted to data of Braithwaite *et al.* (1989).

Indirect changes in structure are usually inferred, since structural change may take a long time to manifest itself (Humphries 1993). Again, there appear to be no good examples that demonstrate threshold damage relationships for the regeneration of native plant species. For example, while Hutchinson and Vankat (1997) documented a negative linear relationship (Figure 5) between cover of the introduced shrub *Lonicera maackii* (Rupr.) Herder and the species richness of native tree seedlings in a deciduous hardwood forest in southwestern Ohio, they found that variability in species richness was high, providing no evidence of a damage threshold.

One instance where a damage threshold could conceivably be demonstrated was in the effect of *Mimosa pigra* L. upon the densities of native tree and shrub seedlings in tropical wetland communities of the Northern Territory (Braithwaite *et al.* 1989) (Figure 6). In this instance, a sigmoidal function could be fitted to the data and a threshold possibly identified. However, the low coefficient of determination for the relationship (r^2=0.33), relative paucity of data, and large gap within the dataset (between approximately 60 and 160 *M. pigra* plants 100 m^{-2}) would certainly not engender confidence in a threshold thus determined!

Major reductions in the capability of infested communities to recover following the removal of weeds have been associated with the crossing of a 'threshold of irreversibility' by Aronson *et al.* (1993), yet another manifestation of the threshold concept. Under conditions where weed infestations are long-standing, the ability of native species to regenerate *in situ* following the removal of such infestations may depend upon the local availability of propagules. In South African fynbos vegetation, Holmes and Cowling (1997) found that many native species regenerated after stands that had been long-invaded (at least 25 years) by *Acacia saligna* (Labill.) Wendl. were cleared. Since these species were not present in the standing vegetation prior to removal of this weed, Holmes and Cowling concluded that persistent seed banks existed for at least some native plants. Lane *et al.* (1997) noted that there were no measurable effects of young stands (2–3 years) of *Mimosa pigra* on the composition, general abundance and patterns of emergence of native herbaceous species present in the soil seed bank of floodplain vegetation in the Northern Territory. Since *M. pigra* had previously been shown to depress the richness and density of the soil seed banks under 10 year-old stands in the same region (unpublished, quoted in Lane *et al.* (1997)), the depletion of seed banks of native herbaceous species by *M. pigra* in the area studied is clearly a damage function that operates over the longer term.

Impact of weeds upon an ecosystem service – water supply. Van Wilgen *et al.* (1996) recently quantified the impact of woody invaders upon catchment water yield in South African fynbos ecosystems. The increased standing biomass associated with the establishment and invasion of alien trees and shrubs resulted in substantial reduction in streamflow, as documented through long-term catchment experiments. The mechanisms of these reductions are not well understood, but reductions are most likely a function of both increased transpiration and interception of rainfall by the invaders. Reductions in streamflow were positively related with above-ground biomass of the vegetation in nine gauged catchments in the Western Cape Province, five of which had been afforested with *Pinus radiata* D.Don to varying extents for different periods (Figure 7). The increase in biomass associated with the establishment and growth of *P. radiata* simply led to an extension of the relationship between the biomass of uninvaded fynbos communities and streamflow. No readily determinable action threshold could be derived from this type of damage relationship.

Multiple invasions. Australian natural ecosystems have commonly experienced invasion by a number of serious weed species, especially in southern Australia (Humphries *et al.* 1991). Whereas the existence of multi-species assemblages of crop weeds has been dealt with by utilizing relative competitive indices (see above), such assemblages generally comprise species that all belong to the same life form. It is considerably more difficult to envisage the use of such indices where different life forms are concerned, having different types of impact upon plant community structure and function. For example, Stockard (1996) describes a subtropical rainforest remnant on the mid-north coast of New South Wales where the canopy had been progressively destroyed through invasion by a shade-tolerant vine

(*Macfadyena unguis-cati* (L.) A.Gentry), followed by a shade-inhibited vine (*Anredera cordifolia* (Ten.) Stennis). On the forest floor a mat of shade-tolerant *Tradescantia albiflora* Kunth up to 60 cm thick prevented the emergence of seedlings of most species. Other serious weeds in this area include the shrubs *Ligustrum* spp. and the invasive tree *Cinnamomum camphora* (L.) Nees (Stockard 1996). It is clear that this remnant will have to be managed in such a way as to minimize its degradation over the longer term. However, it is difficult to imagine how an action threshold could be determined from a damage relationship describing the impacts of the combination of invasive species (with different life forms and life history attributes) threatening the remnant's viability.

Several conclusions may be drawn concerning the damage relationships arising from the presence of weeds in natural ecosystems. Firstly, as with weeds in crops, it will generally be difficult to demonstrate the existence of threshold damage relationships, in large part owing to the large amount of background variation that will likely exist in whatever parameter is being considered. Sample sizes may have to be impracticably large in order to discriminate between the biological threshold damage model and the linear or curvilinear alternatives. Secondly, the form of the damage relationship may vary according to the type of damage under consideration. For example, the invasion of fynbos communities by alien woody plants not only reduces the amount of water supplied by catchments, but also markedly reduces biodiversity and, through increasing fire intensity, exerts further negative effects upon soil erosion and water quality (Le Maitre *et al.* 1996). Whether these relationships can be integrated in order to determine an action threshold is indeed problematic.

One way to obtain an estimate of the integrated impacts of weeds upon biodiversity and ecological functions / services may lie in a better understanding of the inter-relationships of these two facets; there has been considerable discussion in recent years concerning 'what biodiversity is good for' (e.g. Ghilarov 1996, Holdgate 1996, Myers 1996, Bengtsson *et al.* 1997). This has largely stemmed from the need to be able to convince politicians and other decision makers that biodiversity has a variety of direct and indirect values to society, apart from whatever intrinsic worth it might hold. Myers (1996) provides a good summary of the basic key questions, namely 'how does biodiversity generate environmental services; how much biodiversity is needed to do the job; and how far does the relationship depend on local circumstances, especially site conditions (which may change over time)?'

Dynamics of weed populations

Weeds in agricultural ecosystems

For some pests other than weeds, for example fungal pathogens with short-lived spores, there may be little correlation in the size of epidemics in successive years (Wallinga and van Oijen 1997). For these organisms, action thresholds based upon abundance or impact in the current year may be appropriate. However, because most weed populations generate effects that carry over to following years, an economically-defined action threshold that deals solely with impact in the current season is not suited to triggering action on weed control. In these situations the prevention of future population increases should be an additional objective of weed management, and can produce clearly quantifiable benefits. Additional factors, such as weed population dynamics, biology and the stream of benefits and costs attributable to weed control activities, need to be considered. Cousens (1987) termed the action threshold that incorporates these other factors the economic optimum threshold. While it has the strengths of integrating economic and biological considerations over time, a major practical difficulty is the need to characterize fully the population biology of the weed(s) concerned, where numbers may fluctuate unpredictably from year to year. Furthermore, it does not appear possible to calculate a single threshold value that can be used repeatedly, year after year. For these and other reasons, Wallinga and van Oijen (1997) have concluded that the threshold concept does not provide a rational basis for weed control within cropping systems in the long term, lending further support to Cousens' (1987) reservations about the concept. Recent work with annual grass weeds in Australian cereal crops has come to similar conclusions (Medd and Jones 1996, Jones and Medd 1997). These authors found that prophylactic management strategies combining seed kill of wild oats (*Avena* spp.) with other management actions, in order to minimize weed seed bank populations, yielded the greatest long-term economic benefits. Such findings clearly do not support the use of containment management strategies (Cousens 1987), based upon identifiable action thresholds.

An additional, but perhaps most important, concern about the economic analysis associated with the determination of a threshold value is that calculations rarely, if ever, take into consideration the indirect costs that arise from weed control activities (beyond the cost resulting from herbicide damage to the crop). These costs could arise from a variety of effects, including environmental contamination from the use of herbicides, development of herbicide resistance in the targeted species, soil erosion resulting from tillage, etc. It would certainly be difficult to account realistically for such long-term costs in the process of developing an action threshold for weed control in a crop.

Weeds in natural ecosystems

Increases in numbers of weeds may occur both locally and at distance from the site that is being managed, through short- and long-range dispersal, but local increases are the most germane to the consideration of action thresholds at specific sites. For the purposes of this discussion, the establishment of new foci of infestation (Moody and Mack 1988) is treated as an externality, although it clearly is critical to the management of invasions over larger scales. Moreover, this is an aspect that appears not to have been considered in the determination of action thresholds for weeds in agro-ecosystems.

In contrast to the situation with annual weeds in agricultural crops, where the above-ground biomass of the weed populations is reset to zero at certain stages in the cropping cycle, the biomass of perennial weeds of natural ecosystems may increase over substantial periods, as a result of both the growth of individual plants and the increase in plant numbers through recruitment. The impact of these weeds will generally be a function of their collective biomass (e.g. Le Maitre *et al.* 1996).

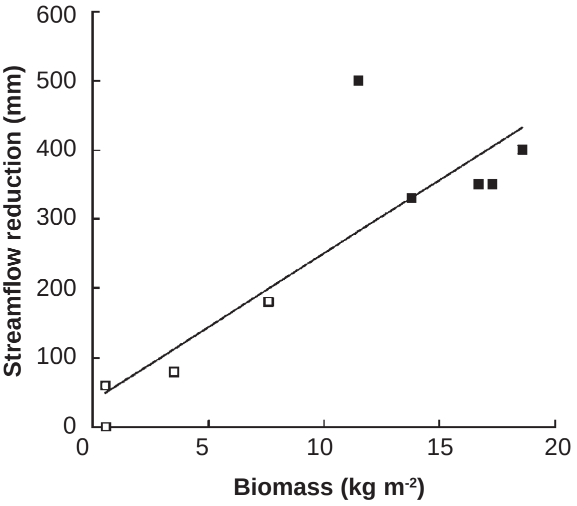

Figure 7. Relationship between reduction in streamflow from nine gauged catchments in the Western Cape and biomass. Fynbos communities were either afforested to varying degrees for different periods with *Pinus radiata* (■) or not afforestated but burned on different cycles (□) ($r^2=0.76, P<0.01$) (from Le Maitre *et al.* 1996).

Table 2. Average work rates (labour days ha^{-1}) under normal conditions for control actions in five infestation densities on study farms in KwaZulu-Natal, South Africa (extracted from Goodall and Naudé 1998).

Habit	Action	Density class (% cover)				
		Maintenance (0–5)	Sparse (6–25)	Moderate (26–50)	Dense (51–75)	Very dense (76–100)
Tree	fell only	3	4	6	9	12
	herbicide^{A}:					
	CS	0.25	0.5	1	2	2
	BS	2	3	5	8	10
	SI	2	3	5	8	10
Shrub	slash only	7	14	24	28	33
	uproot	11	22	37	43	52
	herbicide:					
	CS	1	3	4	5	5
Herb	uproot	6	12	18	30	40
	herbicide:					
	OS	0.25	1	2	3	4

^{A} Herbicide applications comprise CS (cut stump), BS (basal stem), SI (stem injection) and OS (overall spray).

Local (on-site) increase of weed populations may be rapid and relatively predictable; within a single tropical wetland system on the Adelaide River flood plains, Lonsdale (1993) was able to calculate the minimum seed production (68 m^{-2}) required to give stand closure in *Mimosa pigra* infestations within a one year period. This calculation was based upon the observation that a single plant could grow up to 1.8 m tall and produce a canopy area of 1.5 m^{2}. Lonsdale noted a close correlation between rainfall and area increase of *M. pigra* infestations, and attributed this to both increased seedling survival (Lonsdale and Abrecht 1989) and further dispersal of seeds by water in wetter years, but the general picture arising from his studies was that increases in density of sparse populations of *M. pigra* would occur in all but the driest years. Even when climatic conditions are conducive to recruitment, however, the spatial pattern of recruitment will depend strongly upon the distribution of 'safe sites' for regeneration (see Bergelson *et al.* 1993).

This more or less predictable increase in local infestations can be contrasted with situations in which recruitment may be much more sporadic. For example, bursts of recruitment of *Acacia nilotica* ssp. *indica* (Benth.) Brenan in the Mitchell grasslands of north-western Queensland have been related to sequences of wet years (Brown and Carter 1998). In this situation, there could be many years between the establishment of an outlying focus of infestation and subsequent local increase. Phases of increase in weed density are basically unpredictable, although increases in the size of individual plants may be less so.

Costs of control

The increase in size and density of populations of invasive species with the progressive invasion of natural ecosystems means that weed control effort and cost will increase with delays in applying control measures. This is particularly the case where physical control methods are utilized. For example, Goodall and Naudé (1998) found that the labour requirements for slashing or uprooting invasive shrubs or herbaceous species doubled between what they termed a 'maintenance' density (0–5% cover) to a sparse weed density (6–25%) (Table 2). These were large increases in absolute terms, where the basic labour requirements for maintenance densities were, for example, 6 and 11 labour days ha^{-1} for uprooting herbs and shrubs respectively.

Such increases in labour requirements are particularly critical for many environmental weed management efforts in Australasia, where the labour input is often voluntary (Timmins 1995, Leys 1996, Rees and Smith 1996, AACM International 1997). The importance of addressing weed infestations while they can be managed relatively easily and with readily observable results should not be underestimated: 'Once volunteer groups feel overwhelmed by a weed infestation, there are few other management options currently available to address the threat' (AACM International 1997).

Treatment of risk

Weeds in agricultural ecosystems

As stated earlier, weed population dynamics have a large part to play in determining action thresholds in agricultural systems. An understanding of the processes underlying these dynamics will go a long way towards providing realistic predictions of future population levels. However, there remain sources of unpredictable variation in demographic traits, such as the multiplicity of factors affecting seed production, dispersal and persistence (management practices, climatic, biotic and other site characteristics) (Jordan 1992), as well as variations in weed recruitment and survival (Doyle 1991). Population rates of increase may therefore vary considerably from year to year. Further contributors to uncertainty include variations in price for produce, yield response of the crop to the presence of weeds and variations in the efficacy of methods used to control these weeds (Doyle 1991, Pandey and Medd 1991, Pannell 1995). The magnitude of such variation will determine the risk associated with a particular management strategy or action threshold.

For agricultural weeds, risk arises from the chance that unpredictable circumstances will cause immediate and future costs of sub-threshold weed populations to increase substantially beyond expectations. Alternatively, and perhaps less frequently, the actual costs of controlling weed populations arising from the seed production of supra-threshold populations may actually be less than anticipated, thereby indicating that pre-emptive control efforts were greater than required (Jordan 1992). Most farmers are risk-averse to some degree (Auld and Tisdell 1987) and so would be likely to incorporate some 'safety margin' into their determination of an action threshold (Cousens 1987). Furthermore, the tendency to use herbicides prophylactically will be greater with crops of higher value and cheaper herbicides, thus decreasing the relevance of action thresholds (Doyle 1991).

Weeds in natural ecosystems

The uncertainties associated with variations in demographic traits, weed impacts and efficacy of weed control methods that apply to weeds in agro-ecosystems are just as relevant in natural ecosystems. From a practical perspective, however, the major difference between these contexts is that relatively intensive research over the past few decades has provided, comparatively speaking, an abundance of information for weeds in crops. Very little information is available upon which predictions of the population dynamics of environmental weeds can be based. Our ability to predict population growth is at its worst where the 'lag phase' phenomenon (Kowarik 1995) is concerned, i.e. where there is little apparent population growth for a long period, followed by rapid growth. Moreover, recruitment of weeds in natural ecosystems can be highly episodic. It is clear, however, that the potential for local population growth will increase dramatically as colonizing individuals reach reproductive age. The associated risk is that rapid increases in density could make effective control much more difficult. Hiebert (1997) suggested that the urgency of weed control is an important factor in prioritizing weed control efforts. He defined urgency in terms of how much of an increase in effort would be required to achieve successful control, following a delay in action. This approach appears not to have been developed much further, but Goodall and Naudé (1998) presented data relevant to the issue (Table 2).

The degree of risk associated with management decisions in natural ecosystems may be fundamentally greater than with decisions made in agro-ecosystems, in that it may be easier (and cheaper) to get infestations back under control in the latter systems. Nevertheless, if risk is to be dealt with adequately in natural ecosystems, there will be a need to predict the weed population response to control, the physical outcomes of control efforts, and the costs and benefits of these efforts (as well to assess the uncertainties associated with all of the preceding). Without reliable basic information on weed population dynamics, a decision maker is in no position to determine the risks associated with delayed action.

What are the critical decisions and information needs for weed management in natural ecosystems?

Introduced plant species in natural ecosystems vary considerably in terms of their impact; the majority of naturalized species is considered to have only minor ecological effects (Williamson 1996). This suggests that the most important managerial decision in an individual protected area concerns which weeds warrant expenditure on control. The occasional overriding impact of legislation, requiring weed control in protected areas where it might not be warranted on conservation grounds, is acknowledged but not considered further here.

The decision regarding the weed(s) to be targeted should depend upon information, ideally obtained from studies specifically designed to assess weed impact (e.g. Adair and Groves 1998). Panetta and Lane (1996) have argued, however, that in some instances the quite dramatic changes in community composition (i.e. towards a monoculture) and community structure (e.g. from wet grassland to closed forest) rendered by some invasive species may obviate the need to conduct intensive impact assessments. In addition, some of the most damaging invasive species either modify existing disturbance regimes or introduce entirely new disturbances to natural areas (Mack and D'Antonio 1998). These authors maintained that '…alteration of disturbance regime may be the most profound effect that a species or functional group can have on ecosystem structure and function'.

Generally, it is considered that the most cost-effective time to control a weed is during the earliest stage of its invasion (Hobbs and Humphries 1995, Panetta and Lane 1996). As mentioned above, Hiebert (1997) acknowledged the increases in control costs that occur as an invasion progresses. Figures presented by Goodall and Naudé (1998) (Table 2) suggest that the aim of management of serious weeds should be to keep priority areas in the maintenance control phase (less than 5% cover), where a low annual or biennial commitment may suffice to prevent reinfestation, or reinfestation can be reduced by other practices such as burning or controlled grazing.

In practical terms, the progressive increase in effort required to achieve control with increasing degrees of infestation may be a more commanding consideration than weed impact *per se*. Accordingly, the most practically relevant action threshold for managing serious weeds that have already invaded natural ecosystems may be one that is defined in terms of maintenance requirements (Goodall and Naudé 1998). Where the invasion by a serious weed is not sufficiently progressed (or likely to re-occur) in a natural area of high value, eradication may be a feasible objective. The conditions that allow success in an eradication effort are quite restrictive (Moore 1975, Dodd 1990) however, indicating that failure to achieve weed eradication is likely to be the norm. In reality the difference between a failed eradication effort and a successful attempt to confine a weed to the maintenance level (Goodall and Naudé 1998) may not be particularly great.

Conclusions

The most important decisions concerning weeds and natural areas relate to a) determination of the areas most worthy of protection and b) determination of the weeds that are the most threatening to the values of these areas. The importance of weed impact studies lies in their provision of a basis for ranking and prioritizing weeds, rather than in informing decisions on the most appropriate time(s) or weed levels for intervention.

It is telling that, after more than 30 years of research on thresholds and crop damage, almost no farmers use control thresholds (i.e. in the relatively simple systems to which they could be most easily applied) (Cousens 1987 and personal communication). The general failure of control thresholds, based upon measures of weed impact, to deliver benefits in the management of agricultural weeds certainly does not bode well for their applicability to natural ecosystems. Some conclusions may be drawn concerning future endeavours in this area:

i. Benefits provided by natural ecosystems are not easily quantified. There will undoubtedly be controversy over whether such quantification is possible for all of the benefits (as well as how to go about it) for some time to come. This could restrict the scope for the application of cost / benefit analyses in weed management decision-making.

ii. Given the plethora of benefits provided by natural ecosystems, multiple damage relationships need to be determined. Relatively intensive sampling may be required in order to distinguish between alternative forms of damage functions. It is probable that damage relationships demonstrating identifiable threshold levels (equivalent to the commonly-held conception of 'weed management thresholds') will be rarely demonstrable.

iii. The dynamics of weed invasions in natural ecosystems are poorly understood, providing little capability for prediction of either the time course of weed impact or the potential effects of intervention upon this trajectory. As a consequence, there is at present little basis for an informed approach to risk management.

iv. In practical terms, the most meaningful trigger for the management of serious weeds in natural ecosystems may be one that is determined primarily by the cost (and efficacy) of control measures. Managers of natural areas might be best assisted by information on the costs and effectiveness of control efforts, relative to stage of invasion, of their most serious weeds.

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[A] **Footnote**
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