NATIONAL WEEDS PROGRAM

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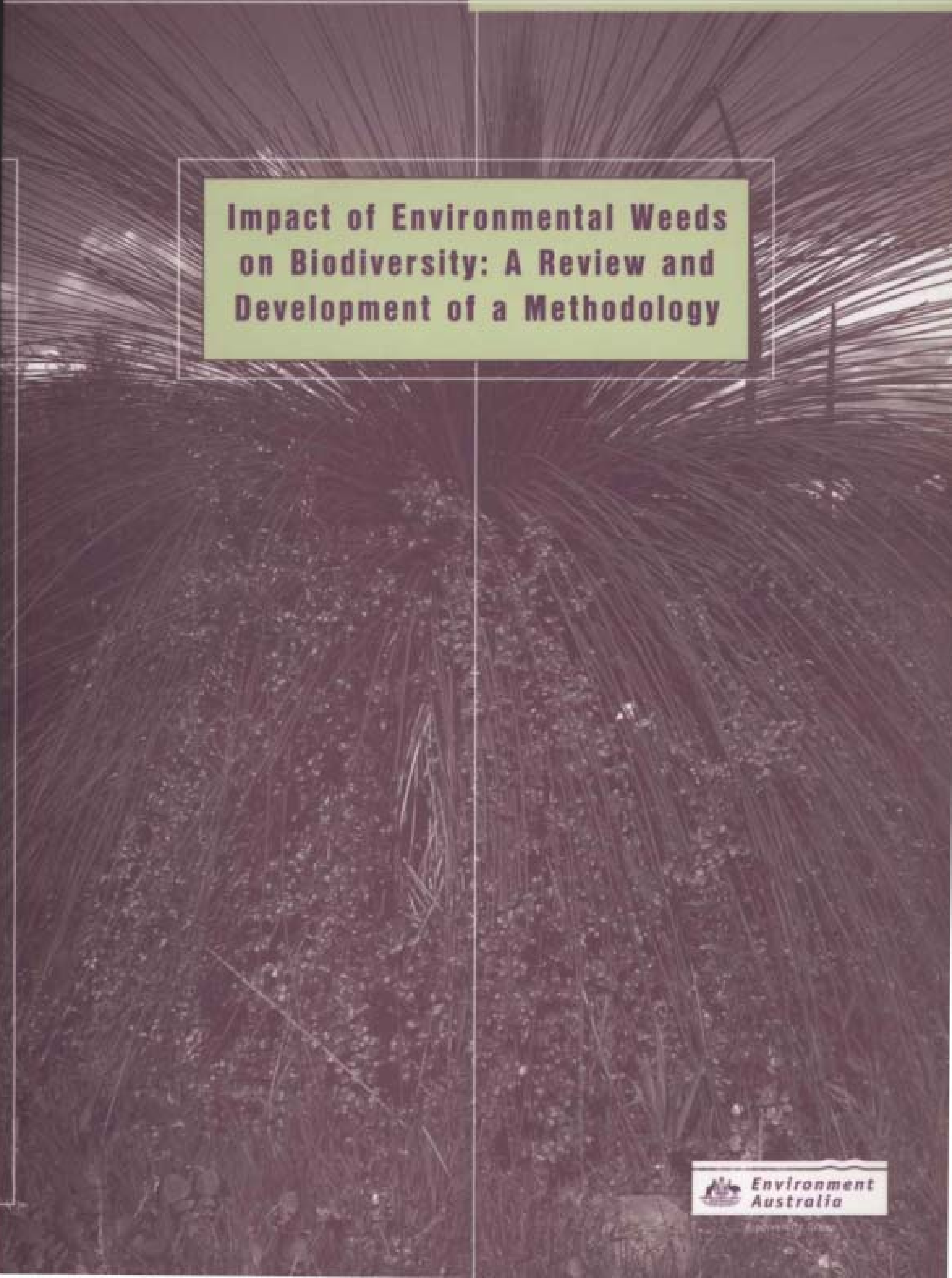

# Impact of Environmental Weeds on Biodiversity: A Review and Development of a Methodology

Environment Australia

**NATIONAL WEEDS PROGRAM**

# IMPACT OF ENVIRONMENTAL WEEDS ON BIODIVERSITY: A REVIEW AND DEVELOPMENT OF A METHODOLOGY

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# FOREWORD

The impact of environmental weeds on the natural environment is an area that has been little studied in Australia. More often than not, impact is assumed rather than quantified. If the impact of environmental weeds on nature conservation values is not quantitatively assessed before a control program is begun, valuable resources can be wasted. For instance, how would it be possible to determine the level at which the weed invasion has a negligible impact on the natural environment? This is a critical question when you are considering investing resources towards the control of environmental weeds.

Key questions that need to be asked before control programs are undertaken for environmental weeds include determining the impact of weeds on native species and ecosystem functions, the threshold densities at which there is little or no impact of the weed on conservation values, and what factors can be manipulated to reduce that impact. This report reviews the impact of environmental weeds on biodiversity values and assesses the techniques used in various quantitative studies. It provides a standardised methodology incorporating experimental design and data analysis options that address the above questions.

This publication is a comprehensive document providing excellent material for future research and planning for environmental weed management. It is an invaluable contribution to the area of impact assessment and I hope it will be considered an important tool for use by all agencies and individuals concerned with biodiversity conservation.

**Gerard Early Acting Head Biodiversity Group**

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# SUMMARY

Environmental weeds threaten nearly all biological communities in Australia. Although weeds appear to degrade many natural ecosystems, quantitative measures of their impact on those systems are relatively rare. Information needed to establish priorities for the control of weeds in natural ecosystems include determination of the mechanisms of weed invasion, the ecological impact of the weeds and the threshold points for declines in biodiversity values as weed invasions proceed. Impact on alpha species-diversity has been the main focus of environmental weed studies to date, both in Australia and overseas. In nearly all cases, the impact of weeds is associated with a decline in native species richness or diversity. Beneficial impacts of weeds were determined only infrequently and mostly occurred on land already degraded, where weeds acted as `nurse plants' for the regeneration of native overstorey components. Where ecosystemlevel functions are altered by weed invasions, habitat conditions or resource availability can be affected adversely for a broad range of species.

Four principal techniques may be utilised to determine weed impact on biodiversity: 1. multi-site comparisons (a quantitative survey technique); 2. weed removal; 3. weed addition; and 4. time-sequence studies. Multi-site comparisons have been widely used, but care is needed in initial site selection. Relatively few studies utilise the other methods, although there are advantages in the adoption of methods that involve manipulative treatments. All four methods have value in measuring the impact of invasive plants and no single method is better than another. Decisions on the time available to undertake weed impact studies and the initial hypotheses being tested will determine which method should be used. Data collection procedures are recommended which focus on the measurement of a broad range of environmental parameters to determine impact of weeds on diversity, ecosystem-level functions and successional consequences.

The impact of weeds on fauna has generally been neglected and needs greater emphasis, as does determination of the threshold levels for declines in faunal biodiversity as well as those for flora. The formulation of management regimes that reduce or prevent further weed invasion and consequent loss of biological diversity is viewed as one of the most urgent tasks for all Australian natural ecosystems.

# INTRODUCTION

## The environmental weed problem in Australia

Plant invasions into natural ecosystems are one of the major threats to the conservation of biological diversity across nearly all biogeographical regions on Earth. It is likely that the invasion of weeds into natural areas has been associated with human movements throughout our evolutionary development, but as is now widely recognised, the rate of this process has accelerated considerably recently. A combination of factors, including developments in transport technology, changes in life-style patterns (particularly in the "western" world), and a seemingly cosmopolitan interest in the introduction and utilisation of foreign plant species, have all been powerful forces in shaping the changing flora of natural areas world-wide. The general trend of an increase in introduced components of natural vegetation accompanied by a decrease in native components has also been greatly assisted by the increasingly extensive exploitation of natural areas, which can alter natural disturbance regimes and thereby provide enhanced opportunities for the colonisation and establishment of introduced plants.

The rate of invasion of the Australian environment by introduced plants has increased linearly since European settlement (Groves 1986), but may now be increasing exponentially in some areas (Carr 1993). Some invasion prior to European settlement probably also occurred, although at a considerably slower rate than at present. For instance, a documented example of a pre-European invasion is that of tamarind *(Tamarindus indicus)* on the northern Australian coastline which is believed to have been introduced by Macassans on their annual fishing expeditions to Australia from about 1700, and certainly prior to the first visit to the region by the European explorer Matthew Flinders in 1803 (Macknight 1976).

Environmental weeds may be defined as those introduced plants which have become naturalised and have invaded natural ecosystems, i.e. ecosystems dominated by native plants that are neither cropped nor routinely grazed by domestic animals. Few, if any, natural ecosystems in Australia appear to be immune from invasion by environmental weeds (Fox & Fox 1986, Carr 1988, Groves 1991). There are, however, considerable differences in the level of invasion, both between and within vegetation formations, with disturbance history and proximity to human developments being key factors that influence the extent and pattern of invasion. The total number of non-native naturalised plants in Australia is about 2000 ( Hnatiuk 1990), although many of these species have not yet been recorded in native vegetation and even fewer are considered to pose a threat to nature conservation values. Swarbrick & Skarratt (1994) listed 1067 environmental weed taxa in Australia, but this number is likely to have increased even since 1994.

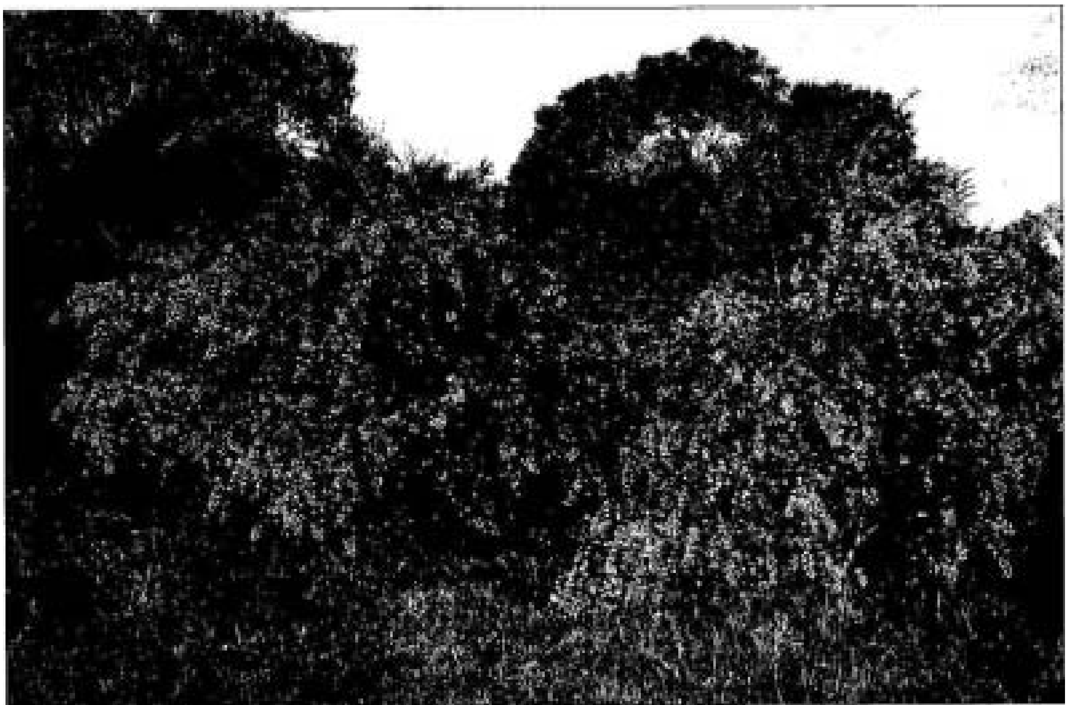

**Figure 1.** **Tagasaste *(Cytisus palmensis),* a nitrogen-fixing shrub used as a perennial fodder plant. Invasions occur in bushland and increased nitrogen levels may promote the incursion of other weeds. A similar situation could occur for other nitrogen-fixing weeds. Photo: R. Adair**

Remarkably few studies have been undertaken to measure the impact of environmental weeds on either biodiversity or ecological functions or even on the processes contributing to invasion, despite the widespread political recognition of environmental weeds as one of the more serious threats to conservation in Australia (e.g., DHAE 1984, COA 1991, BDAC 1992, ESAC 1992, ESDSC 1992, AWC 1993). This discrepancy may have arisen partly because of the obvious negative impact that many environmental weeds have on natural ecosystems. Land managers and research scientists may have been more inclined to invest their effort into implementing weed control strategies rather than in documenting what is already evident as undesirable. In addition, relatively few funds have been available for environmental weed management and research throughout Australia. Such a situation is primarily attributed to a traditional reliance on governmental funding sources, and the lack of a clear connection with industries that have vested interests in removing weeds from areas of conservation importance. It is possible that this situation may change in the future, for instance, as eco-tourism becomes more popular, and as environmental weeds spread further into natural areas. Where environmental weeds also cause damage to economically important industries, as in the case of reduced beef production from Mitchell Grass grassland invaded *by Acacia nilotica,* future control or assessment programs, funded by the relevant industries, may benefit nature conservation ( Adair 1995, Table 2).

There is a clear need to assess the socio-economic impact of environmental weeds in Australia, both in their totality as well as individually for certain of the major invasive species. In this context, it is significant that the one recent analysis of the cost of weeds to the Australian economy ignored the cost of environmental weeds. We still do not know the cost of environmental weeds to Australia nor the attitude and perceptions of the general Australian public to this problem.

**Measuring the impact of environmental weeds - is it necessary?**

Environmental weed invasions are a widespread phenomenon in Australia. Managers of natural ecosystems are required to make decisions on whether or not to implement control programs for particular weeds in areas of significance, and if so, to determine the desired management outcomes and the level of resources required to achieve these outcomes. Inevitable budgetary constraints on weed management programs necessitate that prudent decisions be made on weed control activities, so that the most efficient utilisation of what resources are available are put in place. A range of factors influence decisions on weed control, but the extent of infestations combined with the prospects for achieving management objectives seem to be highly influential. In natural ecosystems, a fundamental issue on which management decisions need to be based is the impact (potential or actual) that a particular weed or a group of weeds has on the long-term conservation of biological diversity. In this respect, the questions that need to be answered are:

(1) What impact are weeds having on the population status of native biota? (2) What impact are weeds having on ecosystem functions within the invaded communities?
(3) What are the threshold levels at which weed invasion has no or minimal impact on conservation objectives?
(4) Are there factors that can be manipulated to reduce the impact and susceptibility of native vegetation to invasion?

Until answers to these questions are considered, let alone found, it is likely that allocation of resources to weed control in natural ecosystems will continue in an *ad hoc* manner and we run the risk of treating the symptoms of invasion rather than the causes.

**Figure 2.**

**Battle of the Monocots! Bridal Creeper climbs a *Xanthorrhoea* in King's Park, Perth. Bridal Creeper is widely established in south-west Western Australia, South Australia and Victoria. Although Bridal Creeper is rated as a serious threat to nature conservation in southern Australia, very little quantitative information is available on the ecological impact of this weed. Photo: R. Adair**

he dilemma that weed managers currently face is that, while many recognise that the above four questions are important, weed impact studies can take several years to complete and therefore the task of finding answers may compete with the often-urgent need to spend resources on `highly visible' on-ground programs for weed control. However, many recent major programs on environmental weed control undertaken in Australia, for instance those against *Mimosa pigra, Chrysanthemoides monilifera* and *Tamarisk aphylla,* have been based on detailed environmental impact studies where the effects of these weeds on biodiversity values were documented and evaluated. Such information is increasingly likely to be a prerequisite for resource allocation to tackle future environmental weed problems in Australia and should thereby provide a sounder basis for the setting of priorities for programs on environmental weed control. Such information also has the dual role of raising both public and political awareness of the importance of environmental weeds, a process which in turn may feed back to influence decisions on the allocation of scarce funds. Perhaps the somewhat controversial views of Anderson (1995) on the impact of the wetland weed *Lythrum salicaria* in North America demonstrate the need for quantitative impact data, for he was unable to locate or collect any quantitative data to suggest that this plant has a negative impact on biodiversity values, despite the very widespread belief that *Lythrum is* causing considerable harm to wetlands in this region.

### Australian ecosystems - are novel assessment methods required?

The simple answer to this question is - No. Weed impact studies have been undertaken across a broad range of vegetation types, from rainforest in Hawaii, to Californian prairie, to South African fynbos and in the sclerophyll forests of south-eastern Australia. In the majority of cases studied to date, the multi-site comparison approach was utilised and clearly demonstrated the potential of weeds to degrade biodiversity values. Australian vegetation associations are highly diverse and contain high levels of endemism, but vegetation floristics or structure do not influence the determination of methodologies for weed impact assessment. More important are statistical considerations and the general principles of experimental design that apply irrespective of the vegetation type under consideration.

### Aim and scope of this report

The aim of this report is to review the impact of environmental weeds on biodiversity in Australian natural ecosystems and to examine and assess the techniques utilised in previous quantitative studies. Anecdotal descriptions of weed invasions have largely been ignored in the preparation of this report, although in many cases such descriptions can provide information that may warrant more detailed and objective assessment. A standardised methodology is then developed that incorporates an experimental design and a data retrieval and analysis process that addresses the key questions often asked about the impact of environmental weeds on biodiversity. The methodology so developed is then applied to two examples. The economic and social consequences of loss of biodiversity due to environmental weed invasions are not considered in detail in this report, although it is recognised that these consequences can be important factors influencing weed management decisions (see above). Assessing the economic and social consequences of environmental weed invasions requires a different methodology from that used to assess impact on biodiversity and is considered to be beyond the scope of this report. Whilst environmental weeds impact on both community and landscape ecology, this report focuses on weed impact at the community scale and does not consider in any detail impacts at the landscape scale, though we acknowledge the significance of the latter.

## Impact of environmental weeds on biodiversity

Biological diversity in natural ecosystems can be considered at three main levels: viz. genetic diversity, species diversity and ecosystem diversity. Many environmental weeds are capable of causing an impact at one or more of these levels, although the degree of impact is rarely quantitatively determined.

#### Genetic diversity

Species can comprise two or more populations each of which may be genetically distinct. Loss of diversity at the genetic level by environmental weed invasions is probably widespread as it includes a reduction in the genetic base of a population at either the local, regional or national scale. Intra-specific genetic variation is the basis for continuing evolution and also the ecological versatility of a species (Adam 1995). Documented losses of populations of native biota are relatively common and, although a number of factors are responsible for these, environmental weeds are often implicated in this process. A large proportion of the published work that measures the impact of environmental weeds reports the loss of species diversity or species richness. These studies focus on specific study sites, but it is reasonable to extrapolate from examples where such losses occur and also where the invasion is widespread to conclude that it is likely that a reduction in genetic diversity of the affected taxon or taxa would occur. With the advent of modern DNA marker techniques such as RAPD, ISA and RAMPS, a reduction in the genetic variability of native biota as a consequence of weed invasion can now be quantitatively determined.

**Figure 3.**

***Marrubium vulgare*** **(Horehound) infestation at Wyperfeld National Park, Victoria. Dense infestations cause substantial degradation of biodiversity values. Removal of Horehound is, however, unlikely to result in the return of indigenous biodiversity without modification of land use practices, viz. grazing and water management. Photo: J. Weiss**

## Species diversity

Species diversity is a relatively well-defined concept and easily measured using standard botanical survey techniques, although there is some debate about the usefulness of species diversity indices. Species conservation is paramount to conservation objectives in natural ecosystems and is the basis on which many decisions concerning management and resource allocation are made. The level of species is the primary focus of evolutionary mechanisms, and therefore species conservation is an important measure in the overall determination of the impact of weeds on biodiversity values. Populations may recover or survive threatening processes (such as weed invasion) that reduce genetic variability, but once a species is lost it is irrecoverable. Surprisingly few taxa appear to have become extinct solely or mainly because of the invasion of weeds. Leigh & Briggs (1992) listed weeds as a major cause in the national extinction of only four plant species: viz. *Gentiana baeuerlenii* (Gentianaceae), *Hydatella leptogyne* (Hydatellaceae), *Hypsela sessiliflora* (Lobeliaceae), and *Trachymene scapigera* (Apiaceae). However, the inevitable expansion of the current weed flora, and the arrival of new environmental weeds in the future, are likely to increase the rate of species extinction through direct competition and disruption to essential ecosystem functions and structure. In this regard, Leigh & Briggs (1992) listed 57 plant species that are nationally endangered as a result of competition with weeds, whilst in Victoria 166 native plant taxa have been listed as threatened by environmental weeds (Carr *et al.* 1992). In Victoria, 68 species of plant or animal and 12 vegetation or animal communities listed under the *Flora and Fauna Guarantee Act* (1988) have weeds identified as a significant threat to their survival (see Appendix).

There are many anecdotal accounts of environmental weeds affecting native plant or animal populations. Summarised details of studies that quantify weed impact are given in Table 2. Nineteen of the 20 studies on environmental weed impact undertaken in Australia demonstrated a decline in either species richness, canopy cover or frequency of native species. Only one study - that on *Lycium ferocissimum* (Erkelenz 1993) - failed to detect any adverse effect (Table 2). In two cases, *Pittosporum undulatum* (Mullett & Simmons 1995) and *Cytisus scoparius* (Waterhouse 1986), species richness was reduced by 47% and 59% respectively, and the cover/abundance of the surviving species was also substantially lowered. In most cases direct comparisons of the impact of weeds on biodiversity values are not possible because of differences in procedures for data collection and presentation. Nearly all overseas studies also reported a negative effect of weed invasion on some measure of biodiversity, usually species richness and/or percentage canopy cover; only Williams (1983), Knof & Olsen (1984) and De Pietri (1992) reported beneficial roles of environmental weeds, whilst Smathers & Gardner (1979), Mueller-Dombois & Whiteaker (1990), and Anderson (1995) found no negative effects associated with invaders.

Whilst most of the Australian studies have focused on the impact of a particular weed species, others have concentrated on determining the environmental or land-use factors associated with weed invasions. The results of these studies provide useful insights into the mechanisms of weed invasion; furthermore, they can highlight areas where differences in land management alter the susceptibility of native vegetation to invasions and can provide an assessment of the effect of various environmental factors on broad-scale alpha diversity levels. McIntyre & Lavorel (1994) provided a detailed analysis of invasions into grassy woodlands in New South Wales; in their study, the impact of environment on the diversity of native species was separated from that arising from disturbance as a result of weed invasion.

The majority of impact studies undertaken on environmental weeds, both in Australia and elsewhere, focused on changes in floristic communities and ignored the impact of weeds on faunal assemblages. A range of effects on fauna have been reported. Most studies indicate substantial declines in species richness and abundance of both vertebrates (Winterbottom

*1970,* Brock *et al. 1986,* Braithwaite *et al. 1989,* Griffin *et al. 1989)* and invertebrates ( Slobodchikoff & Doyen *1977,* Samways & Moore *1991).* Several notable descriptive accounts indicate that the impact of weeds on faunal communities is widespread and possibly severe ( Macdonald *et al. 1986,* Copley *1988,* Usher *1988,* Drake *et al. 1989,* Lane *1992).* The effects of weeds on fauna can be complex or problematic in terms of land management. The promotion of native fauna through weed invasion does not necessarily aid native flora. In New Zealand, for instance, invasion of the aquatic herb *Lagarosiphon major* has lead to increased grazing by swans and crayfish which have contributed to the decline of native aquatic plants (Howard-Williams & Davies *1988).* Conversely, weeds may encourage survival of exotic fauna by provision of food and shelter. A greater emphasis on assessment of the impact of weeds on faunal communities is needed.

**Figure 4.** ***Anthoxanthum odoratum*** **(Sweet Vernal Grass) is invasive in a wide range of habitats in southern Australia, particularly soils with impeded drainage. It is extremely difficult to control. Exotic grasses can alter ecosystem functions thereby causing substantial and often irreversible changes to native biotic communities. Photo: R. Adair**

## Ecosystem diversity

Ecosystem diversity is a measure of ecosystem types, the diversity of habitats and of the ecological processes occurring within them. Adam (1995) correctly points out that there is no consensus on the appropriate level for the analysis of ecosystem diversity. Broad-scale divisions may reflect major physiognomic groups (e.g. grassland, tropical rainforest) and can be made with little difficulty, but there are no accepted taxonomies for units at a finer scale. Where environmental weeds threaten Australian ecosystems, it is usually where the invasive species creates a new stratum and/or where essential ecosystem functions are altered. The vigorous Madagascan liana *Cryptostegia grandiflora* (Rubber Vine) threatens the species-rich native vine thickets, gallery forests and dry rainforest in the monsoonal belt of northern Australia ( Humphries *et al.* 1991) by smothering existing overstoreys. *Acacia nilotica* (Prickly Acacia), on the other hand, threatens Mitchell Grass grasslands by creating a new and woody overstorey stratum. On tidal mud flats in south-eastern Australia, the maritime grasses *Spartina* spp. (Cord Grass) threaten internationally significant communities of wading birds (Lane 1992) by altering geomorphological processes, especially through increased siltation, which eventually leads to the displacement of critical feeding areas. As a further example, *Cenchrus ciliaris* (Buffel Grass) threatens plant and animal communities in central Australia, mainly by increasing the intensity and frequency of natural fire regimes (Latz 1991).

Most accounts of the impact of environmental weeds focus on the population or species level and describe changes in species richness, diversity, fitness, abundance, and frequency. Whilst these changes could have occurred as a result of ecosystem-level changes, unless essential environmental parameters are measured such changes are not detected. Many weed invasions into natural ecosystems may not, however, cause functional changes (Vitousek 1990). In such cases, environmental weeds may reduce or displace native components with the invasive species taking the place of its native counterparts in a functional sense. The invasions of *Stipa neesiana* into temperate *Themeda triandra* grassland, and of *Aponogeton distachyon* into southern waterways are perhaps examples of this process. Several environmental weeds are reported to cause shifts in successional pathways as a result of changed environmental conditions. Such shifts are expected to alter the floristic composition and structure of vegetation, thereby degrading biodiversity values. Successional shifts usually occur over considerable periods of time (decades cf. years) and can be difficult to detect unless survey techniques are employed to specifically measure such changes. These have been undertaken for some woody weeds, such as *Cytisus scoparius* (Waterhouse 1986, Smith 1994), *Pittosporum undulatum* (Mullett & Simmons 1995), *Hakea sericea* (Williams 1992a), *Hakea salicifolia* (Williams 1992a) and *Ulex europaeus* ( Lee *et al.* 1986). Environmental weeds that are capable of altering ecosystem functions alter the living conditions for all species within the affected communities and therefore are likely to lead to substantial declines in species diversity. Such weeds require intensive management and should be a higher priority for control over those that may displace native species without changing ecosystem-level functions meaningfully (Vitousek & Walker 1989). A complete assessment of the impact of environmental weeds on biodiversity, therefore, should also measure the status of critical ecological processes, such as native seedling recruitment, nutrient cycling, hydrological and geomorphological processes, disturbance regimes and plant-animal interactions.

There are relatively few Australian cases where environmental weeds have altered ecosystem functions (Table 1). In reality, the number of environmental weeds that affect ecosystem functions is likely to be considerably larger than that reported in the literature. This low number probably reflects the low level of resource allocation to this type of research. Outside Australia, quantitative studies on the ecosystem-level impact of environmental weeds are also rarely reported (see Table 2 for examples).

**Table 1. Environmental weeds known to affect ecosystem-level functions**

**A) Australia**

| Species | Altered function | Source |
|---|---|---|
| *Cenchrus ciliaris* | Increased fire intensity and frequency | Latz (1991) |
| *Cytisus scoparius* | Alters successional pathways, fire intensity | Waterhouse (1986), Smith (1994) |
| *Ehrharta calycina* | Increased fire frequency | Baird (1977) |
| *Pinus spp.* | Altered hydrological patterns, physical and nutritional properties of soils altered | NFAC (1983) - particularly Pilgrim *et al.* (1982), Hamilton (1965) |
| *Pittosporum undulatum* | Decreased fire frequency | Mullett & Simmons (1995) |
| *Tamarisk aphylla* | Altered trophic structure of avifauna and reduced habitat availability for reptiles. | Griffin *et al.* (1989) |
| *Mimosa pigra* | Converts aquatic grassland/sedgelands to shrublands | Braithwaite *et al.* (1989) |
| *Spartina spp.* | Converts tidal mudflats to grassland by altering geomorphological processes | Lane (1992) |

**(B) Overseas**

| Species | Country | Altered function | Source |
|---|---|---|---|
| *Acacia, Hakea, Pinus* | South Africa | Altered fire behaviour, altered spatial distribution of nutrients, hydrological patterns | van Wilgen & Richardson (1985),Witkowski & Mitchell (1987), Stock & Allsopp (1992) |
| *Andropogon virginicus* | USA, (Hawaii) | Lower soil evaporation leading to accelerated erosion | Mueller-Dombois (1973) |
| *Myrica faya* | USA, (Hawaii) | Increased levels of available nitrogen | Mueller-Dombois & Whiteaker (1990), Vitousek &Walker (1989), Smathers & Gardner (1979) |
| *Lonicera japonica* | USA | Inhibits reproduction of overstorey dominants in mixed deciduous forest | Thomas (1980) |
| *Lupinus arboreus* | USA | Increased nitrogen levels | Maron & Connors (1996) |
| *Eichhornia crassipes* | USA | Alters water chemistry | Ultsch (1973) |
| *Psidium cattleianum, Ligustrum robustum* | Mauritius | Regeneration of overstorey native trees suppressed. | Lorence & Sussman (1988) |
| *Tamarisk spp.* | USA | Increased salinity levels, lowered water tables, altered stream flow and flooding regimen | Graf (1978), Loope *et al.* (1988) |
| *Rhododendron ponticum* | Ireland | Regeneration prospects of shrubs and trees | Cross (1982) |
| Poaceae | Worldwide | Increased fire frequency, higher rates of nutrient loss, altered microclimate, prevention or delay of succession | D'Antonio & Vitousek (1992) |

Table 2. Quantitative studies determining the impact of weeds on biodiversity in natural ecosystems in (A) Australia and (B) Overseas.

A) Australia

| **Weed** | **Source** | **Community** | State | **Method'** | **Impact** | **Impact** measure [2] | **Statistical** Analysis | **Sample** size | **Sample unit area (m)** (dimensions) | **Comments** |
|---|---|---|---|---|---|---|---|---|---|---|
| *Asparagus asparagoides* | Sorensen & Jusaitis (1995) | Mallee shrub | SA | *M* | *Asparagus* invasion appears to have reduced the density of *Pterostylis arenicola* | N | N/A | 31 | 1 (1x1) | Temporal monitoring plots were established in front of an advancing population of *Asparagus*. Preliminary results suggest there is a negative impact of *Asparagus* on *Pterostylis* density. Longer monitoring is required. |
| *Chrysanthernoides monilifera* | Weiss & Noble (1984) | Coastal dunes | NSW | MC | Displaced the ecological analogue *Acacia longifolia,* reduced seed production in *Acacia* | S, %CC, NR | Pearson's correlation matrix, step-wise multiple regression | 84 | 25 (5x5) | Alternative hypotheses tested to confirm invasion *of Chrysanthemoides* as the primary factor causing changes in native vegetation. |
| *Cytisus scoparius* | Smith (1994) | Eucalypt open-forest & woodland | NSW | MC | Reduced species richness, altered successional pathways | S, %CC | Kruskall-Wallis one-way ANOVA | 190 | 1 (1x1) | Invasion results in conversion to more mesic conditions, thereby altering successional pathways. |
| *Cytisus scoparius* | Waterhouse (1986) | Eucalypt open-forest & woodland | NSW | MC | Reduced species richness, reduced % ground cover, reduced tree regeneration | S, %CC, N | Paired plots, chi-square test, Mann-Whitney U-test | 28 | 25 (5x5) | Succession scenarios presented. Successionary pathways could be altered *by Cytisus* invasion. |
| General | Cale & Hobbs (1991) | Roadside mallee | WA | MC | Reduced species diversity | H' (Abundance measure not listed) | t-tests, multiple regression, correlation matrices | 390 | 1 (1x1) | Correlation between exotics and soil phosphorus levels. High phosphorus levels linked to weed invasion. |
| General | Hester & Hobbs (1992) | Shrubland & woodland | WA | R | Reduced % cover of natives, reduced seed production | N, %CC | t-tests | 48<br>403 | 1 (1x1)<br>0.25(0.5x0.5) | Removal of weeds resulted in 3-fold increase in native cover. |
| General | McIntyre *et al.* (1988) | Wetlands | NSW | MC | Reduced species richness | S | Principal component analysis, correlation matrices- Spearman correlation coefficent, regression | 179 | 50 (varied) | Negative correlation between number of exotic species and number of native species. |

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|---|---|---|---|---|---|---|---|---|---|---|
| General | McIntyre & Lavorel (1994) | Grassy woodland | NSW | MC | Reduced species richness | S | General linear modeling, log-linear modeling | 120 | 30 (6x5) | Examines vegetation response to environmental disturbance & land use. Negative correlation between exotic & rare native species. Methods separate impact of disturbance and presence of exotics. |
| General | McIntyre (1993) | Grassy woodland | NSW | MC | Reduced species richness | S, %CC | Chi-square test | 120 | 30 (6x5) | Examines invasion patterns in communities with low or high species richness. |
| General | Milberg & Lamont (1995) | Sclerophyll woodland | WA | MC | Reduced % cover of natives | S, %CC | Descriptive, contingency tables, chi-square test | 60 | 2 (2x1) | Examines impact of fire on invasion. Difficult to separate effects of fire from that of weed invasion. |
| General | Bridgewater & Backshall (1981) | *Eucalyptus & Banksia* woodland | WA | MC | Species diversity declined with exotic invasions | S, B, H' | N/A | 24 | 0.25 | Suggested that positive feed-back loops may occur between fire and the invasion of annuals. Suggested that disturbance-adapted indigenous plants be utilised in revegetation programs in nature reserves. |
| *Hypochoeris radicata* | Gilfedder & Kirkpatrick (1993) | Semi-natural grassland | TAS | L | Reduced rate of growth of a native species | B | | | | *Hypochoeris* suppresses *Helipterum*. Field experiments are needed to verify laboratory findings. |
| *Lantana camara* | Fensham *et al.* (1994) | Dry rainforest, savanna woodland | QLD | MC | Reduced species richness, reduced % canopy cover | S, %CC, N | Spearman's rank correlation coefficients, Mann-Whitney U-test | 102 | 100 (10x10) | Invasion of *Lantana* thought to be linked to pig damage which opens tree canopy. *Lantana* results in increased fuel levels which when burnt can kill canopy trees. |
| *Lycium ferocissimum* | Erkelenz (1993) | Coastal | SA | MC | No evidence for displacement of native plants | S, %CC | t-tests, linear regression | 52 | 4 (2x2) | No detectable impact on native species. |
| *Mesembryanthemum crystallinum* | Kloot (1983) | Annual pasture | SA | R | Reduced biomass production of annuals | S, B | N/A | 10 | 4 (2x2) | |
| *Mimosa pigra* | Braithwaite *et al.* (1989) | Tropical wetland | NT | MC | Reduced species richness, reduced density of tree seedlings | S, %CC, N, B | Multiple regression, correlation matrices | 14<br>70 | 1100 (55x20)<br>1 (Ix1) | Abundance of birds and reptiles reduced. Mammals and amphibians were not adversely affected. One small mammal was favoured by *Mimosa* invasion. |
| *Pinus pinaster* | Kermode (1993) | Heath | VIC | MC | Reduced native species richness, cover and abundance | S, %CC | Descriptive | 62 | 8 (2x4) | A belt transect passed through invaded and uninvaded heath. In invaded heath species richness was reduced to 31 % of nearby uninvaded heath. Pine height and basal area showed an inverse relationship with the number of heath species. |

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| *Pinus radiata* | Friend (1982) | Eucalyptus open forest | VIC | MC | Reduced species richness of native fauna. Higher proportion of introduced fauna | S, %CC, D | Chi-square | 70 | N/A | Seventy habitat surveys were undertaken. Several survey methods were used to record mammals: trapping, scat analysis, spotlight survey, signs and diurnal surveys. Age of the *Pinus* plantation influenced mammalian diversity. |
| *Pittosporum undulatum* | Mullett & Simmons (1995) | Dry sclerophyll forest | VIC | MC | Reduced species richness, reduced % cover of natives | S, %CC | Multivariate pattern analysis, two-way table analysis, ordination, correlation matrices | 52 | 6 (3x3) | Light reduction by heavy canopy cover was believed to be an important factor in the decline of native biodiversity. There were indications of invasions altering successional pathways. |
| *Pittosporum undulatum* | Gleadow & Ashton (1981) | Eucalypt open-forest | VIC | MC | Reduced species richness | F | Pattern analysis, ANOVA | 186 | 4 (4x4) | Heavy shading suppressed native species. Higher levels of some nutrients occurred beneath canopies. |
| *Tamarisk aphylla* | Griffin *et al.* (1989) | Riparian | NT | MC | Altered species evenness, reduced diversity of birds & reptiles | S, F | Chi-square test | 594 | 25 (10x2.5) | Frequently distribution of native plants was altered, but there was no measurable change in species richness. Richness of reptiles and birds was reduced. |

1. Method: R=Removal of weed, A=Addition of weed, L=Laboratory studies, M=Temporal monitoring, MC=Multi-site comparisons (usually weed presence cfweed absence), S=Succession study, P=Predictive
2. Impact measure: S=Species richness (number of species), %CC=Percentage canopy cover, N=Number of individuals in the sample unit, NR=Number of reproductive units in the sample, H'=Shannon-Weiner index, F=Species frequency, E=Evenness, B=Biomass, N/A=Not applicable (B) Overseas

| Weed | Source | **Community** | Country | f Iethod' | Impact | **Impact , Measure** | **Statistical Analysis** | **Sam** Ie size (N) | **Sample unit area (m)** | **Comment** |
|---|---|---|---|---|---|---|---|---|---|---|
| *Alliaria petiolata* | McCarthy (1997) | Flood plain forest | USA | R | Removal of *Alliaria* resulted in increased cover of indigenous annuals, herbaceous and woody vines and tree seedlings | %CC, B, S,H' | Linear regression, ordination using detrended correspondence analysis, ANOVA, log-linear modeling | 9 | 1 (circular) | Removal experiments were conducted over 3 years. Removal of *Alliaria* stimulated regeneration of annuals, herbaceous and woody vines and tree seedlings. The impact on slower growing perennial herbs, grasses, sedges and shrubs was unclear. Analysis techniques are skillfully applied to data. |
| *Ammophila arenaria* | Slobodchikoff & Doyen (1977) | Coastal dunes | USA | MC | Sand-burrowing arthropods and numbers and diversity decreased as marram grass cover increased | %CC,S,H | Correlation analysis | 24 sites | 100m long transects | Use *of Ammophila arenaria* to control movement of coastal sand dunes seems unjustified in areas with nature conservation values. |

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| *Ammophila arenaria* | Boyd (1992) | Coastal dunes | USA | MC | Positive and negative associations between *Ammophila* and other plant species were found | %CC, F | Chi-square test, Association analysis | 42 (6 onset s a 7 s as) | 17 m strip transects. 1 (1x1) | Four plant species were negatively associated, 3 species were positively associated and 3 species were not influenced *by Ammophila. Ammophila* may provide habitat for rodents which forage on *Cakile*. Microdistributions along line transects reveal trends that may otherwise be missed. |
| *Ammophila arenaria* | Barbour *et al.* (1976) | Beach vegetation | USA | MC | Reduced cover diversity and evenness | %CC, H', E | Linear regression | 34 | strip transects (1x40-100m) | *Ammophila* was negatively correlated (P=95% or 99%) with species cover diversity and evenness of species cover. The authors modified the Shannon- Weaver species diversity index to utilise species cover rather than species density. |
| *Baccharis pilularis* | Hobbs & Mooney (1986) | Annual grassland | USA, ( California) | S | Reduced % cover, biomass & seed production of herbaceous species | S, B, NR | Descriptive | 25 | 0.25 (0.5x0.5) | Impact was dependent on the age class of *Baccharis. Baccharis is* a native shrub, but the methods may be suitable for weeds if age classes can be recognised. |
| *Brachypodium pinnatum* | Bobbink & Willems (1987) | Chalk grassland | UK | MC | Reduced species diversity | H', B | Wilcoxon sign test, Mann-Whitney U-test | 37 | 0.03 (0.3x0.3) | Determined the critical above-ground phytomass of *Brachypodium*. Phytomass was used as the abundance component in the Shannon-Weiner index. *Brachypodium is* native to chalk grassland. |
| *Buddleja davidii* | Smale (1990) | Alluvial stream beds | New Zealand | S | Displaces primary native colonisers, facilitates the establishment of other woody shrubs, ferns, mosses & sedges | %CC, S, age classes, N | Correlation coefficients, logistic regression | 56 | 4 (2x2) | *Buddleja* displaced native primary colonisers and accelerates successions to forests on fresh alluvium. In areas subject to frequent flooding and alluviation, *Buddleja may* persist indefinitely. |
| *Campylopus introflexus* | Equihua & Usher (1993) | Heather | UK | L | Reduced germination rate *of Calluna* | | | | | The impact of *Campylopus* under field conditions is extrapolated using laboratory results. |
| *Centaurea maculosa* | Tyser & Key (1988) | Fescue grasslands | USA | M | Reduced species richness, reduced species abundance | S, F | Linear regression, two-way ANOVA, Newman-Keul's test, contingency tables, Fisher's extract test | 137 | 0.01 (0.2x0.5) | A decline in species richness was correlated to *Centaurea* density. Changes in frequency patterns of a native species were detected over a 4 yr period. |

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| *Centaurea maculosa* | Rice *et al.* (1992) | Native grassland Soft-wood forest | USA | R | Removal of *Centaurea* had either no measurable impact on biodiversity or resulted in a slight increase | %CC, F, H' | Descriptive | 12 (3 epl ca es as 4 sees) | 1120-acre | Herbicide treatments were used to remove *Centaurea.* Initial declines in species diversity were slight and treated areas had slightly higher diversity levels 2.5 yr after treatment. Study over a longer time period may have detected more pronounced trends. Initial infestations of *Centaurea* were low. |
| *Centaurea maculosa, Phleum pratense* | Tyser (1992) | Fescue grasslands | USA | MC | Reduced species richness, reduced species diversity, reduced % canopy cover | S, %CC, H' | Kruskal-Wallis one-way ANOVA | 80 | 0.01 (0.2x0.5) | A negative impact on cryptogram crust was detected. Ecosystem-level impact suggested. Canopy cover was used as an abundance measure for H'. Disturbance is likely to be important but disturbance indices were not collected. |
| *Cyathea cooperi* | Medeiros *et al.* (1992) | Rainforest | USA (Hawaii) | MC | Reduced ground cover | %CC | Descriptive | 12 | 400 (20x20) | Dense stands of *Cyathea* were reported to suppress understorey species diversity & biomass, but no data was provided. |
| *Cytisus scoparius* | Williams (1983) | Successional scrub | New Zealand | S | On disturbed areas, facilitates re-establishment of native trees | N, %CC, F | Descriptive | 20 | 40 (20x2) 200 (20x10) | Broom was replaced by native broad-leaf trees after about 15 years. |
| *Dipsacus sylvestris* | Huenneke & Thompson (1995) | Riparian | USA | P | Teasel has the potential to compete with a rare *Cirsium* | N/A | N/A | N/A | N/A | The potential of *Dipsacus* to degrade populations of a rare *Cirsium* was determined by habitat descriptions, germination response, demographic comparisons & competition experiments. |
| *Elaeagnus angustifolia* | Knof & Olson (1984) | Riparian | USA (Rocky Mountains) | MC | Increased habitat availability to riparian bird species | H', E | Descriptive | | | *Elaeagnus* altered horizontal and vertical structure of vegetation. Birds using tall-scrub were provided with additional habitat. Impact on plant diversity was not assessed. |
| *Euphorbia esula* | Belcher & Wilson (1989) | Mixed-grass prairie | Canada | MC | Frequency of 5 dominant native grasses reduced, species richness and species diversity reduced | F, H', S, B | Correlation coefficients | 50 | 0.16 (0.4x0.4) | Cover values of flora were determined by point quadrats. Frequency data were used for correlation determination. Five grass species declined with *Euphorbia* frequency. Disturbance favours *Euphorbia* establishment. |

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| General | Reader & Bricker (1994) | Deciduous forest | Canada | M | Disturbance increased invasion | S | Two-way ANOVA, Tukey's HSD test | 68 | 1 (1x1)<br>2 (2x1) | Determined the threshold values for logging intensity and plot size on invasion by non-forest species. Invasion was monitored over 3 years. |
| *Hakea salicifolia* | Williams (1992a) | Scrub | New Zealand | S | Replaced by natives with time | S, %CC, N | Descriptive | 21 | 40 (10x4) | *Hakea,* an early succession species, was replaced by natives after 60 yr. Soil type was an important influence on succession outcomes. There were no signs of replacement by natives on thin soils. |
| *Hakea sericea* | Williams (1992b) | Scrub | New Zealand | S | Replaced by natives with time | S, %CC, N | Descriptive | 10 | 50 (10x5) | Degraded herbaceous vegetation. In Scrub, *Hakea is* replaced by natives. |
| *Hedera helix* | Thomas (1980) | Mixed deciduous forest | USA | R | Suppressed understorey herbs | F, B, %CC, growth rate | t-tests | 20 | 1 (1x1) | Removal of *Hedera* promotes regeneration of understorey herbs. Woody plant regeneration was apparently not suppressed. Impact varied according to habitat. |
| *Hieracium sp.* | Treskonova (1991) | Alpine grasslands | New Zealand | M | Change in grassland structure and composition | %CC | Canonical correspondence analysis, correspondence analysis | 53 | 100 (N/A) | Degradation of native tussock grassland is linked to invasion of *Hieracium,* particularly *H. pilosella.* A network of interrelated influences appear to govern this trend. The analysis methods demonstrate a statistical approach to interpretation of changes in cover values over time. |
| *Lagarosiphon major* | Howard-Williams & Davies (1988) | Aquatic herbfield | New Zealand | MC | Displaced native vegetation. Promotes faunal changes which also contribute to pressure on native plants | S, %CC,B, N | Multiple regression, correlation coefficients, descriptive | 42 | 0.25 (circular) Line transects | The number of native aquatic species decreased as height and biomass of *Lagarosiphon* increased. Large beds of *Lagarosiphon* attract swans and crayfish which adversely affect native plants. |
| *Lonicera japonica* | Thomas (1980) | Mixed deciduous forest | USA | R | Suppressed understorey herbs and regeneration of woody plants including dominant overstorey species | F, B, %CC, growth rate | t-tests | 19 | 1 (lxl) | *Lonicera* suppressed the regeneration of herbaceous and woody plants, including overstorey dominants. Light availability limited *Lonicera* growth and impact. Impact varied according to habitat. |

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| *Lonicera tatarica* | Woods (1997) | Dry-mesic forest | USA | MC | Total herbaceous cover, herb species richness and density of tree seedlings are depressed | S, %CC, N | ANOVA, linear regression, detrended correspondence analysis, multidimensional scaling | 304 | 4 (2x2) | Four sites were sampled. Data from a dry site were different from that of wetter sites. Ordination indicates that evergreen herbaceous species and sprawling or vining species may be more tolerant *of Lonicera* suggesting its impact may be related to seasonal competition for light. Ring counts of tree seedlings show inhibition of recruitment post invasion. |
| *Lupinass arboreus* | Maron & Connors (1996) | Coastal prairie | USA | MC | Reduced species richness. Increased nitrogen levels promoted invasion by exotic grasses and thistles | S, B, N | Kruskall-Wallis Test, Mann-Whitney U-test, one -way ANOVA, linear regression | 16<br><br>159 | 0.09 (0.3x0.3)<br><br>1 (1x1) | *Lupinus* promoted weed invasion by increasing nitrogen levels & creating bare ground. Thistle number and size increased in association with *Lupinus. Lupinus is* native to Californian prairie. |
| *Lythrum salicaria* | Anderson (1995) | Wetland | USA | MC | No evidence of species declines detected as a result of invasion | S, %CC, D | ? Correlation statistics Descriptive | 9 | 1250 (50x25) | Although *Lythrum* caused changes in the appearance of wetlands, no quantitative data were available to suggest that biodiversity values are affected. Sampling at 1 site showed there was no correlation between species richness and *Lythrum* canopy cover or density. Seventy-nine papers were reviewed and all lacked data showing negative impacts on biodiversity. |
| *Mesembryanthemasm crystallinum* | Vivrcttc & Muller (1977) | Coastal grassland | USA | M | Reduced grassland species richness, density, biomass, species' stature. Survivors were later to flower and seed set | S, %CC, D, B | Descriptive | 20 | 0.63 (loom radius circle) for quantitative measures. A *6xro* plot with multiple line tratnects for invasion patterns. | *Mesembryanthemum* invasion was measured over 3 yr. The species established in unoccupied or sparsely occupied areas, and then grew into surrounding areas. Few grassland seedlings established beneath dried *Mesennbryanthemum*. A differential seedling establishment pattern was correlated with high levels of salt found in soil beneath dried *Mesenabsyanthemum*. Osmotic interference appears to be the invasion mechanism. |

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| *Myrica faya* | Mueller-Dombois & Whiteaker (1990) | Lava flow | USA (Hawaii) | MC | Weeds not found to be associated with *Myrica* invasion | S | Two-way table | 25 | 100 (circle with ,adios 5.64m) | Suggested that a growth period of 10 yr may be insufficient for invasion trends to be detected. *Myrica* was preferentially ssociated with *Buddleja*. |
| *Myrica faya* | Vitousek & Walker (1989) | Lava flow | USA (Hawaii) | MC | *Myrica* increased level of biologically available N | Nitrogen levels | One -way ANOVA, Newman-Keul's test, Scheffes multiple contrasts | | | *Myrica* increased N availability via N Fixation. Suggested that successional processes could be altered. Determined that N was limiting primary production and that N input from *Myrica* was available to other organisms. |
| *Myrica faya* | Smothers & Gardner (1979) | Lava flow | USA (Hawaii) | M | No negative impact detected | S, %CC, N, F | Descriptive | 18 | 100 (10x10) 200 (10x20) | Ohia trees provide microsite conditions for establishment of *Myrica*, but competition for water reduced vigour *of Myrica*. |
| *Myriophyllum spicatum* | Madsen *et al.* (1991) | Aquatic herbfield | USA | M | Diversity of native submerged macrophytes decreased with advancing *Myriophyllum* | %CC, S, F | Descriptive | N/A | A central plot, 6x6 with sub quadrats, and 4 radiating transeres each 25 m | Over 3 years, *Myriophyllum* progressively increased in abundance and reduced total species diversity from 20 to 9 species. Mean diversity declined from 5.5 species per quadrat to 2. Invasion occurred in the absence of detectable disturbance. High biomass production and the formation of a dense upper canopy that reduced light intensity is a possible invasion mechanism. |
| *Pinus spp.* | Cowling *et al.* (1976) | Fynbos | South Africa | MC | Reduced species richness, simplified vegetation structure | S, %CC | Descriptive | 55 | 200 (10x20) 25 (5x5) | The Braun-Blanquet method of vegetation description and classification was used to determine the status of understorey vegetation of *Pinus* stands. Lacking in strong quantitative data but impact was apparent. |
| *Pinus spp.* | Richardson & van Wilgen (1986) | Fynbos | South Africa | A | Reduced species richness, % CC and density | S, %CC, N | Descriptive | 393 | 0.1 (0.4x0.25) | In heavily invaded sites, the return to pre-invasion state after the removal of weeds was unlikely due to elimination of some groups of species. |
| *Pinus, Cupressus* | Samways & Moore (1991) | Grassland | South Africa (Kwa-Zulu, Natal) | MC | Reduced richness & diversity of grasshoppers, reduced richness of grasses | Range of diversity indices | ANOVA, correlation matrices | 62 | 5 (5x1) | Recreational shade trees influence grasshopper assemblages. Cypress have no detectable impact, but Pines have a negative impact by shading feeding areas. |

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| *Psidium catteianum, Ligustrum robustum* | Lorence & Sussman (1988) | Tropical rainforest | Mauritius | MC | Reduced species richness, reduced regeneration of native trees | S, N | Descriptive | 10<br>10 | 50m line transect.<br>4 (2x2) | Sampling seedling numbers in undisturbed forest provided early warning of ecological collapse. |
| *Rhododendron ponticum* | Cross (1982) | Oak woodland, heath | Ireland | MC | Reduced species richness and abundance, ecosystem-level alterations suggested | S, %CC | Descriptive | 20<br>16 | 1 (1x1)<br>1-25 (N/A) | Impact was largely due to light reduction. There was a negative impact on epiphytes, fungi, animals and regeneration of shrubs & trees. |
| *Rosa rubiginosa* | De Pietri (1992) | Subantarctic forests | Argentina | MC | *Rosa* behaves as nurse crop for natives | S, %CC, N | Regression, ANOVA | 1 | 47x40 | *Rosa* shortened recovery succession after disturbance by providing protective micro-environments. |
| *Sapium sebiferum* | Bruce *et al.* (1995) | Texas coastal prairie | USA | S | Vegetation structure altered. Graminoids and forbs replaced with trees and shrubs | S, D, %CC | Kruskal-Wallis test | 76 | 250 (circular) | The chronosequence method was used. Study stands were categorised into age classes. Aging was based on ring counts. Dominance of life forms shifted rapidly (<10 yr) as graminoids and forbs were replaced by trees and shrubs during succession. Woody species showed significant increase in density with stand age suggesting that woodlands may become more diverse. There was no indication of reversion back to prairie. |
| *Tamarix chinensis* | Hunter *et al.* (1988) | Riparian woodland | USA | MC | Impact of *Tamarisk* on avian fauna varied according to location | S, D | Friedman's two-way ANOVA, descriptive | 28 | Variable distance transects | Use of *Tamarisk by* birds along the Pecos River was high amongst all groups in all seasons. This was in contrast with avian use in 2 other valleys. Elevational (climate) gradients are suggested reasons for this phenomenon. Analysis of data on a seasonal use and feeding-group basis provided useful insights on impact of *Tamarisk*. |
| *Tradescantia fluminensis* | Kelly & Skipworth (1984) | Lowland forest | New Zealand | MC | Regeneration of native tree seedlings inhibited | B, N (tree seedlings) | Linear regression | 31 | 0.82 (0.91x0.91) | Data were collected from a small forest remnant. Thick mats of *Tradescantia (up* to 60 cm) appeared to suppress native tree regeneration. Light intensity was positively correlated with *Tradescantia* biomass. |

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| *Tribulus terrestris* | El Ghareeb (1991) | Abandoned fields | Kuwait | MC | Flowering delayed in natives, density & biomass of natives reduced | S, N, B | t-test | 200 | 0.25 (0.5x0.5) | |
| *Ulex europaeus* | Lee *et al.* (1986) | Podocarp -dicotylous forest | New Zealand | S | Native plant regeneration suppressed | S, %CC, N, F | Descriptive, correlation matrices | 125 | 4 (2x2)<br>100 (10xlO) | Estimated that native canopy would return in 50-60 yr as gorse canopy cover declined. Protection from fire or mechanical disturbance was needed for this process to occur. |
| Various. Mostly grasses | Wester (1994) | Herbfield | Hawaii | M/R | Weeds reduce %CC in dry years | S, %CC, F | ANOVA, Mann-Whitney U-test | 53<br>6 | 100 (10x10)<br>1 (1x1) | Weed invasion was linked to climatic conditions. Weed control was unnecessary for *Marsilea* protection as populations re-established with onset of normal rainfall. |

1. Method: R=Removal of weed, A=Addition of weed, L=Laboratory studies, M=Temporal monitoring, MC=Multi-site comparisons (usually weed presence cfweed absence), S=Succession study, P=Predictive

2. Impact measure: S=Species richness (number of species), %CC=Percentage canopy cover, N=Number of individuals in the sample unit, NR=Number of reproductive units in the sample, H'=Shannon-Weiner index, ti F=Species frequency, E=Evenness, B=Biomass, N/A=Not applicable

## Techniques for assessing the impact of environmental weeds on biodiversity

Determining the overall impact of environmental weeds requires consideration of five main factors:

1. the range of habitats susceptible to invasion;

2. the rate of spread within and between susceptible habitats;

3. the prospects for the containment or control of the

weed;

4. the relationship between the weed and habitat disturbance; and

5. the impact of the weed on species diversity.

These points could form the basis of a rating system for environmental weeds in Australia by developing a series of scoring levels within each of the points listed above. Recently, Hiebert (1997) proposed such a ranking system based on similar criteria, while Macdonald & Jarman (1984) also applied a similar approach to rate invasive weeds within the fynbos biome of South Africa. Whilst a rating system based on the range of susceptible habitats, rate of spread, prospects for control, relationship to disturbance and impact on biodiversity may be useful for objectively scoring weeds already present in Australia, it is inadequate for determining the potential impact of new weed incursions or of those not yet present in the country. The screening procedure proposed by the Australian Weeds Committee (1994) is far more powerful in this respect as biogeographical/historical and biological/ecological data are incorporated into a pre-entry risk assessment.

This section reviews the methodology suitable for determining the impact of weeds on species diversity within an invaded habitat. Thus, we focus on alpha diversity (species diversity within a habitat) but, clearly, knowledge on the range of habitats that are susceptible to invasion is an important prerequisite for measuring impact on gamma diversity. Samson & Knop (1984), although not dealing specifically with weed invasion, clearly place the need for assessments of both beta diversity (species diversity between habitats) and gamma diversity (species diversity within a region) within a clear conservation context.

### Species diversity - assessment approaches

*Multi-site comparisons.* Research on the impact of environmental weeds has been directed primarily towards determining the effect of weeds on alpha diversity. The different methods utilised are listed in Table 3. The majority of studies (38 out of 61, Table 2) utilised the "multi-site comparison" approach (= chronosequence studies of Woods (1997)) where sample units are located randomly within a particular habitat or vegetation formation at sites with the weed present and in comparable areas where the weed is absent (control). No manipulative techniques are employed. Sites are generally located within a relatively discrete study area. On occasions, however, considerable distances separate weed-free sites from those invaded by weeds, thereby reducing the suitability of this approach. The main advantage with this method is that detailed data can be collected within a short period of time (weeks or months), thereby providing a rapid assessment of weed impact. An underlying assumption with this method is that the species composition in invaded areas was the same as or similar to control areas prior to invasion. As pre-invasion states cannot be determined, considerable care needs to be taken to match control and weed-invaded sites and to explore alternative hypotheses that could explain differences in the data collected. Weiss & Noble (1984), working on *Chrysanthemoides,* and Maron & Connors (1996), evaluating the impact of *Lupinus,* provide good examples where testing of alternative hypotheses has been used. In both cases, alternative analyses supported the original hypothesis that differences in biological parameters between invaded and control sites were most likely caused by the weed rather than other factors. Analyses of data from multi-site comparisons are mostly based on correlation statistics and as such lack the power of manipulative studies, such as the weed addition or removal approach. While multisite comparisons are useful in detecting impact trends, they are most appropriate when manipulative studies have already been undertaken (Morrison 1997) and a negative impact demonstrated or where they are combined synchronously with manipulative studies.

**Figure 5.**

**Burnt Mallee vegetation near Meningie, South Australia. Removal of regenerating *Asparagus asparagoides* (Bridal Creeper) from replicated plots enables the impact of this weed on native species to be measured when comparisons are made with control areas, i.e. the "removal technique". Photo: R. Adair**

***Weed removal and addition studies.*** This approach, termed "neighbour manipulation" (Aarssen & Epp 1990) has been used widely to measure competition effects between a broad range of mostly non-weedy plants. Where an environmental weed has reached its biogeographical limits with extensive "infilling", it can be difficult to find suitable control sites where the weed is absent or at low densities. In such situations, the manipulative approach of removing the weed from discrete areas and monitoring subsequent vegetation changes in comparison with matched weedy areas can provide a measure of weed impact. Hester & Hobbs (1992) used the "weed-removal" technique to measure the impact of weeds in remnant vegetation in Western Australia. Kloot (1983) used this technique to determine the impact of *Mesembryanthemum crystallinum* in annual pasture. In both cases, the presence of weeds reduced species richness, reproductive fitness and/or biomass. In Hawaii, Wester (1994) used the removal technique to

**Table 3. Main assessment approaches used in determining impact of environmental weeds**

| Approach | Advantages | Disadvantages |
|---|---|---|
| Multi-site comparison | • Rapid data collection<br>• Suitable for regional studies<br>• Large sample sizes possible<br>• Only practical method for assessing impact on fauna<br>• Can identify the impact of the invader on successional processes and invasion models (as per Luken 1997) within a short time period | • The condition and history of sites prior to invasion is usually unknown, thus invasion impacts remain speculative<br>• Uninvaded control sites can be difficult to find for widespread species |
| Weed removal | • Strong evidence for impact<br>• Suitable for widespread infestations<br>• Suitable for species studies or weed communities<br>• Applicable to recent weed invasions | • Labour intensive<br>• Replicate size is usually low due to high maintenance levels<br>• Weeding methods can affect diversity<br>• Long-term studies required<br>• Not suited to old invasions where the invader has caused irrecoverable damage to ecosystem functions or biodiversity |
| Weed addition | • Pre-invasion state can be measured<br>• Strongest evidence on impact<br>• Control over density levels<br>• Chronological studies possible<br>• Useful for weeds with limited distribution e.g. new arrivals | • Long-term studies required<br>• Risk of `escapes' into native vegetation<br>• Unpopular with land managers |
| Time sequence | • Pre-invasion state can be measured<br>• Chronological studies possible | • Long-term studies required<br>• Actively spreading populations required<br>• Control sites can be difficult to find if the invader is widespread |

measure the impact of weeds on the rare fern *Marsilea villosa* and found that *Marsilea* was more responsive to weather conditions than to weed competition. In an area recently burnt by wildfire, Virtue (pers. comm. 1996) is using the removal approach to measure the impact of *Asparagus asparagoides* (Bridal Creeper) on post-fire succession of mallee vegetation in South Australia. As fire can stimulate the regeneration of many native plants, post-fire conditions can offer excellent opportunities for determining the impact of weeds on the reproductive success of native species. In this situation, weed-removal techniques are very appropriate as control sites that have been exposed to the same fire conditions, but lack weeds, may be difficult to locate.

Pimm (1987), El-Ghareeb (1991) and Gilfedder & Kirkpatrick (1993) also refer to the use of " weed-removal" methods. In most cases, careful hand removal of the invader has been adopted, but in the case of Rice *et al*. (1992), herbicides were utilised to selectively remove *Centaurea* from native grassland in the USA. While the herbicide treatments had a slight short-term negative effect on associated dicots, *Centaurea* was effectively removed. This approach may be applicable to any weed where selective chemical removal techniques are available, e.g. Bitou Bush on coastal dunes in Australia (Toth *et al.* 1996), but a possible impact of chemical residues needs to be taken into consideration. Aarssen & Epp (1990) elaborate on other deficiencies with the plant removal method.

The weed-removal technique is not suitable for measuring impact in situations where the presence of the weed(s) permanently prevents the recovery of native species to a pre-invasion state. Such a failure is likely to occur in vegetation types dominated by obligate re-seeders with short-lived soil-seed banks where dispersal from neighbouring areas is unlikely. Unless removal experiments are conducted prior to the widespread establishment of the invader, the experiment will measure the residual effect of the invader, rather than an unaltered community (Walker & Smith 1997) and, as McCarthy (1997) found, will tend to measure the response of taxa with strong dispersal or vegetative reproductive abilities. Where the functional properties of the habitat or ecosystem have been irreparably disrupted, the removal technique does identify successionary pathways that could occur with large scale treatment programs. This is well demonstrated with the removal of *Pteridium* (Bracken) in Britain where efforts to restore species-rich grass heath and heather heath were thwarted by grass invasion after Bracken removal, possibly due to increased inputs of atmospheric nitrogen into the ecosystem (Marrs & Lowday 1992, Pakeman & Marrs 1992). As the weed removal technique depends on recovery over considerable periods of time (years or even decades) this technique is generally of limited use in assessing weed impact. However, Thomas (1980) detected a negative impact on native plant regeneration by the exotic lianes *Lonicera japonica* and *Hedera helix* in mixed deciduous forest in USA within 1.25 yr, indicating that in some habitats data can be collected within a reasonable period of time.

The weed removal technique is most applicable in assessing the impact of new arrivals, in either a spatial or temporal sense, before ecological damage occurs and is certainly the most appropriate technique for evaluating weed impact on rare or endangered species. In relation to endangered taxa, while it may be tempting to remove weeds from all populations and monitor recovery processes, control sites where weeds are left intact are required to verify impact. There may be instances where weeds protect or enhance endangered species and their removal will not result in recovery, e.g. partial dependence of the Eastern Barred Bandicoot *(Perameles gunniz)* on *Ulex europaeus* for shelter in western Victoria. Where detection of rare species is required in sampling procedures, the proposals of Green & Young (1993) for determining sample sizes are useful.

The "weed-addition" approach, where weeds are introduced into a previously uninvaded site, also requires long monitoring times to assess impact, but can provide irrefutable evidence on the impact of weeds in natural ecosystems as pre-invasion states can be measured accurately. Additionally, density levels of the plant invader can be controlled permitting examination of density-dependent effects and determination of threshold levels of invasion impacts. This technique has its greatest value in determining the impact of newly arrived weed species as control sites can be readily established and replication levels for addition treatments are restrained more by time and resources than by habitat availability, which can seriously limit evaluation procedures using other methodologies. While it is common for weed addition experiments to be carried out under artificial conditions e.g. pots in a glasshouse, very few attempts have been made to use this technique under field conditions, but it is there that meaningful ecological data are collected. Apart from time, the most significant disadvantage of the "weed-addition" method is that weeds can be difficult to contain and thus, there is a risk of weeds escaping into native vegetation outside the experimental area. This problem can be avoided by undertaking experiments a safe distance from areas of high conservation value. Richardson & van Wilgen *(1986)* utilised the weed-addition technique to conclude that the introduction of *Pinus* spp. decreased species diversity in fynbos vegetation after 35 years. Although the addition technique has rarely been applied to assess the impact of invasive plants, the method has considerable potential and warrants wider application.

***Time sequence studies.*** Another approach used to determine weed impact is "time-sequence" studies where the movement and impact of specific weed populations are monitored over time. Pre-invasion states can be determined ahead of an advancing population but, in the few cases where this technique has been employed (e.g. Madsen *et al. 1991,* Tyser & Key *1988,* Smathers & Gardner *1979),* monitoring over considerable periods of time is required and there is no certainty that changes will be detected within the observation period. Where historical survey data are available, repeated surveys in the same area are able to demonstrate changes in floristic composition. Providing data on environmental variables are collected, the relationship between vegetation changes and environmental influences can be interpreted using ordination methods. Treskonova *(1991)* provides an interpretation of changes in releve data from alpine grassland in New Zealand over *26-28* years using such techniques and implicates the invasion of *Hieracium* with the loss of indigenous plant biodiversity.

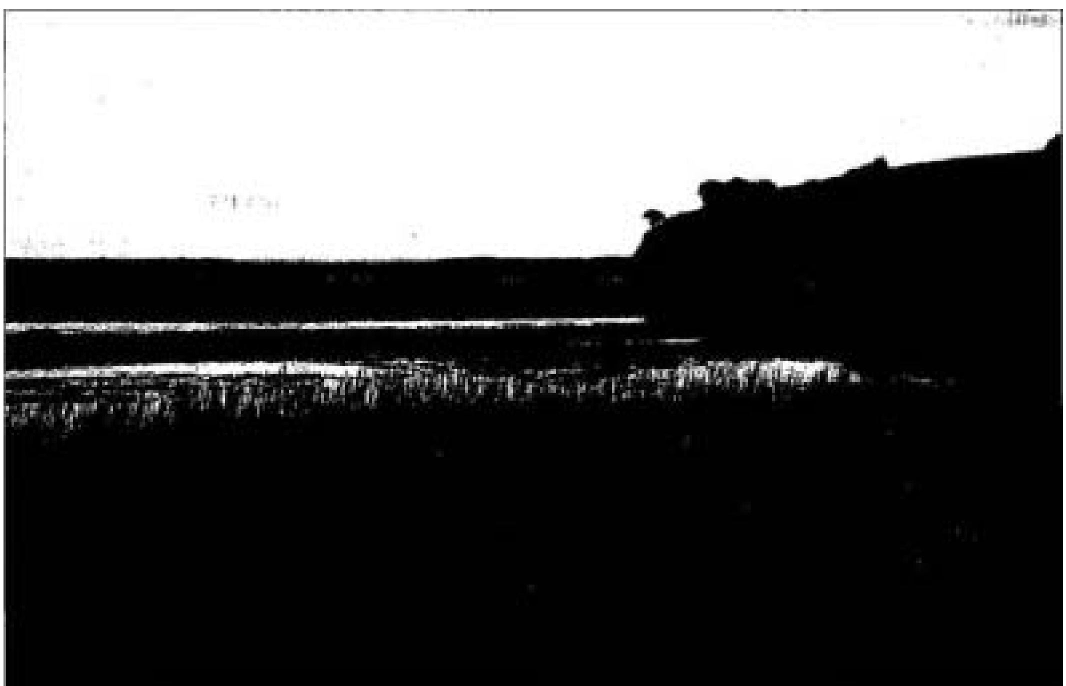


**Figure 6.**

***Spartina* (Cord Grass) on a tidal mud flat used as a feeding ground by migratory wading birds. *Spartina* is a relatively new weed in southern Australia and threatens significant bird populations by converting mud flats into grassland, thus causing major changes to ecosystem processes. Photo: K. Blood**

## Ecosystem-level functions

Studies aimed at determining the impact of weeds on ecosystem-level functions invariably use the multi-site comparison approach with expanded data collection procedures that measure key functional processes (Tables 1 & 2). Manipulative experiments are also often employed to measure biological responses to changed habitat conditions. For example, Vitousek & Walker (1989), working on the impact of the nitrogen-fixing shrub *Myrica* in Hawaii, measured changes in nitrogen levels within stands of *Myrica.* They ascertained whether nitrogen was a limiting resource within the invaded habitat, and then determined whether the nitrogen fixed *by Myrica* was available to other organisms. Complementary studies by Mueller-Dombois & Whiteaker (1990) and Smathers & Gardner (1979) attempted to measure changes in vegetation as a result of *Myrica* invasion. Similarly, manipulative experiments were used by Maron & Connors (1996) to test the hypothesis that increased nitrogen levels associated with the establishment of *Lupinus* are leading to the invasion of exotic grasses and herbs in a Californian prairie. The multi-site comparison approach has also been used to identify the relationship between land management practices and weed invasion, as well as the impact of weeds on biological diversity. McIntyre *et al*. (1988), McIntyre (1993) and particularly McIntyre & Lavorel (1994) identified environmental factors linked to weed invasion and species diversity in grassy woodland in northern New South Wales, and Treskonova (1991) used similar techniques in alpine grassland in New Zealand. McIntyre & Lavorel (1994), using generalised linear modeling techniques, were able to identify and distinguish the impact of weed invasion and disturbance on species diversity. As Woods (1997) correctly states, distinguishing the impact of invasion from disturbance events which can lead to invasion is not easily achieved, but is important in designing control strategies for invasive weeds. Where disturbance events promote invasion, treatment of weeds rather than invasion stimulus can be of little avail.

Clearly, the data collected from survey sites determine the types of analyses and conclusions that can be drawn from studies of weed impact. The questions that were originally asked about the problem will determine what data are collected and how they are collected. In terms of developing a standard methodology for evaluating the impact of weeds on biodiversity values, the basic requirements for data collection are to determine the impact of weeds on alpha diversity levels, the processes by which weeds may affect diversity levels and, importantly, whether ecosystem-level alterations occur as a result of weed invasion. Subsequent work should then focus on determining the impact of weeds on both gamma diversity levels and on threshold points where diversity values are affected adversely.

Data collection for determining ecosystem-level changes will and should vary enormously depending on the species under study and the mechanisms of disruption involved. Vitousek (1990) identified three means by which exotic species (plants or animals) may alter ecosystems: viz. (1) by causing changes in resource availability; (2) by altering the trophic structure of an area; and *(3) by* changing disturbance frequency or intensity. A single mechanism or, more likely, various combinations of these may occur. We do not believe it possible or desirable to develop a standardised recommendation on which parameters should be measured for determining ecosystem-level changes as the number of possibilities is large. Instead, a standardised approach would, at the very least, propose a hypothesis on ecosystem-level impact and endeavour to test the hypothesis using a range of data collection methods.

Evaluation of the successional consequences of environmental weed invasions places weed impact within a temporal context and can yield meaningful results not readily detected by studies focusing on a narrow range within a successional sere. Luken (1997) is perhaps the first to provide an ecological overview of non-indigenous invaders in this context. Williams (1983, 1992a,b) studied two weeds, *Hakea* and *Cytisus* (Scotch Broom), on previously cleared forest sites in New Zealand, and found that these weeds were eventually replaced by native overstorey species and native species richness increased, thus fitting the `temporary dominance' model of Luken (1997). *Cytisus* is considered to be a desirable species where the re-establishment of native forest is the aim of vegetation management (Williams 1983). In Australian sclerophyll forests, however, *Cytisus* appears to alter successional processes by creating more mesic conditions and favouring species more aligned to cool temperate rainforest (Smith 1994). Similarly, the coastal tussock grass *Ammophila arenaria,* although it appears to cause declines in biodiversity in beach vegetation (Barbour *et al.* 1976), may in the longer term facilitate the invasion and establishment of native woody components that eventually lead to the displacement of *Ammophila* (Bergin *et al.* 1997). Quantitative successional studies could be used to clarify the impact of *Ammophila* on dune biodiversity. Determining the age of invasive plants or populations is essential for successionary studies and dendro-chronology (Bruce *et al.* 1995, Williams 1983), size (Hobbs & Mooney 1986) and fruit condition (Williams 1992a,b) have all been used to age stands of invading species.

Sampling for weed impact within a successional context can clarify disturbance/invasion relationships and the time period over which these may occur. Such information is relevant to priority setting for management of environmental weeds, particularly if the biodiversity implications are also considered in the sampling process. In a management context, priority must be given to those taxa that fit the `long-term dominance' model (Luken 1997) where the invader assumes dominance and maintains this role for long periods of time as a result of competitive interaction, regeneration or changed disturbance. Taxa that fit into the `temporary co-existence' model, where the invader does not assume dominance and fails to persist, or the `temporary dominance' model (Luken 1997) are of lesser ecological importance.

***Gamma*** *diversity*. There are no easy methods for determining the impact of environmental weeds on gamma diversity and there are no published accounts dealing with its assessment available. A systematic determination of weed impact on gamma diversity would require data sets on:

1. the impact of the invader on alpha diversity levels across the main habitat types within the weed's range;
2. the potential bioclimatic range of the weed;
3. the distribution of susceptible habitats within the potential bioclimatic range of the weed; and
4. the distribution of land management practices or environmental triggers that pre-empt or promote invasion.

By overlaying these data sets with the assistance of software packages, such as "GIS" ( Geographic Information System), the impact of an environmental weed within a regional context could be more readily appreciated. The use of "GIS"-type programs in mapping potential weed impact is not a new concept and has recently been utilised by Everitt *et al.* (1995) for *Euphorbia esula* (Leafy Spurge). Further exploration into the feasibility of using this approach is required before recommendations on its adoption can be made.

***Fauna.*** Ecologists have largely neglected determining the impact of weeds on faunal communities. In Australia, both Braithwaite *et al.* (1989) and Griffin *et al.* (1989) determined the impact of weeds on fauna by applying standard capture or observation techniques for each of the faunal groups under examination in both weed-invaded and comparable weed-free areas. This approach of multi-site comparison, so frequently utilised for measuring impact on vegetation, is probably the only practical technique available for determining weed impact on the majority of faunal assemblages. For avifauna, while interval point counts at different times of the day over a 3-5 day period were used to measure impact within a relatively short time period (Pyke & Recher 1984), longer-term studies extending over several seasons and analysed according to feeding-group and seasonal-use, e.g. Hunter *et al.* (1988) seem to provide more powerful insights into weed impact. For mammals, reptiles, some amphibians and ground-dwelling invertebrates, specifically designed trap lines replicated throughout the study area may be suitable, but systematic searches can also provide valuable additional data. Multiple survey methods, e.g. Friend (1982), are more likely to reveal fauna diversity than the use of a single survey method. Systematic visual or auditory surveys can be utilised for amphibians, while beating, sweeping or trapping can be employed for arthropods. Soil fauna, although less visible than other faunal components, should also receive attention in weed impact studies. Changes in nutrient cycling due to weed invasion are ultimately the direct result of alterations to microbial and soil arthropod assemblages. Techniques for assessing soil fauna are well established and readily applicable to weed impact studies. Unlike the more mobile aboveground, mega-fauna, sampling for changes in soil faunal communities can be achieved with relatively small sample areas.

Site matching and the critical habitat size of specific faunal groups require careful consideration in weed impact studies. Site comparison should only be made between areas that are closely matched on enviro-physical criteria and if potentially large enough to support the fauna group under study. For example, there would be little point in comparing the avifauna of a 5 ha plot of weed-infested vegetation with large tracts of uninvaded vegetation when the critical habitat size for most components of that habitat exceeds 5 ha. Clearly, false conclusions on weed impact would occur as habitat size would be a confounding factor. A requirement for large habitat patches can constrain the number of sampling opportunities for evaluating weed impact on fauna, thereby increasing the potential for pseudoreplication (Hurlbert 1984). Where this situation occurs, examination of distribution trends along spatial gradients maybe of more use analytically.

### Data quality and presentation

Diversity measures are widely used for monitoring the impact of environmental pollution, including weed invasion, and there has been considerable discussion evaluating the suitability of various techniques. Magurran (1988) provided a comprehensive and practical review of diversity measures which was largely used in this report.

Whilst a broad range of diversity measures are available, relatively few have been utilised to assess weed impact. In most cases, the simple measure of species richness - the number of species within a unit area - is used as the primary indicator of diversity. It is easily calculated and meaningful in ecological terms. Thirty-two out of the 43 studies listed in Table 2 utilised this measure. Other measures of species richness are available and listed succinctly by Stocker *et al.* (1985), but have rarely been adopted by weed ecologists. As Magurran (1988) correctly pointed out, although species richness is an informative measure, considerable ecological insights can be gleaned from more detailed investigation into the variety and abundance of species and from shifts in abundance distributions or dominance patterns. Measures of abundance are also commonly reported in weed impact studies with a variety of parameters being quantified. Percentage canopy cover, as visual estimates, cover scales, cover/abundance rating (e.g. Braun-Blanquet cover values), or quantitative measures (point quadrats) are also used widely and provide useful indicators of impact. However, as considerable differences in estimates of visual cover can occur between recorders, objective quantitative methods are preferable. Additionally, as cover-scale estimates are not linearly correlated with abundance, biased results can be created if these data are used in diversity models or used as an abundance measure in diversity indices (Magurran 1988).

The number of units (genets or ramets) of a particular species within a sampling unit (N) is particularly useful information for assessing weed impact as it is a direct measure of abundance. Few studies on environmental weeds, however, collect such information, probably because of the time required in counting taxa with numerous numbers of small individuals. In many cases workers have selected an ecologically important component of the habitat under study and concentrated abundance measures on these. For example, counts of the number of regenerating over-storey species (usually trees) provides a measure of the impact of weeds on the regeneration capabilities of a functionally important component of the invaded habitat and therefore facilitates an assessment of whether ecosystem-level changes may be occurring. Braithwaite *et al.* (1989), Hester & Hobbs (1992) and Fensham *et al.* (1994) all provide good examples of this type of data collection. Biomass measures are also useful in some situations for measuring abundance and, although tedious, provide objective abundance data that can be incorporated into diversity indices. The frequency of occurrence - the number of sampling units within which a species occurs - has occasionally been used successfully ( Williams 1983, Tyser & Key 1988, Griffin 1993, Wester 1994, and Mullett & Simmons 1995), but needs to be used with discretion. It can lead to an under-estimate of the abundance of the commonest species as there is no density component in this statistic (Magurran 1988), e.g. 50 quadrats, each with one individual of a species, will rank equally with those that have 100 individuals in 50 quadrats. Hengeveld (1979) listed 14 widely differing definitions of abundance and provides a caveat about the interpretation of abundance data.

Diversity indices have been infrequently used in weed impact studies with Cale & Hobbs (1991), Bobbink & Willems (1987), Tyser (1992) and Knof & Olson (1984) using the widely criticised, but popular, Shannon-Weiner Index of diversity. The Shannon-Weiner Index is based on the proportional abundance of a species and is a collective measure of evenness and species richness. In most cases the Shannon-Weiner Index utilises density data for component species, but Barbour *et al.* (1976) modified the index to utilise canopy cover values. The Shannon-Weiner Index and others that incorporate dominance values can provide insights into the impact of weeds additional to measures of species richness on their own. However, Hurlbert (1971) strongly condemned the use of such indices and suggested instead that "species composition parameters" having straightforward biological interpretations be presented as alternatives to the diversity (indices) approach. This approach seems to have been adopted subsequently by weed impact ecologists. Magurran (1988) and Stocker *et al.* (1985) provided comprehensible accounts of the various indices available. Magurran (1988) suggested that species diversity indices simpler than the Shannon-Weiner Index be utilised and recommended the alpha log series index as an alternative, together with the Margalef and Berger-Parker indices as quick measures of abundance and dominance components of diversity. These indices are simple to calculate, easy to interpret and statistically and ecologically sound.

In situations where weed invasion has caused severe biological degradation, virtually any measure is likely to detect changes in biological diversity. On the other hand, in situations where the impact is more subtle, more sensitive techniques may be required to determine significant differences between invaded and uninvaded areas. It is better to plan for the latter than to face the risk of investing time in data collection only to subsequently find that the methods employed eventuated in Type II experimental errors (i.e. accept the original hypothesis [there are no differences between invaded & uninvaded areas] when the alternative hypothesis [differences exist due to weed invasion] is true). In the early stages of invasion or with weakly invasive plants, changes in the shape of species abundance distributions may occur before detectable differences in species richness can be found. Impact analyses should therefore attempt to find evidence for changes in both species richness and species abundance distributions. The latter have been largely neglected by weed ecologists, although Griffin *et al.* (1989) used frequency data in a similar manner to demonstrate the impact of *Tamarisk* on riparian vegetation in central Australia.

Sampling methods for determining the impact of environmental weeds follow the procedures that apply to biological studies in general and can be found in most standard texts on ecological methods, such as Kershaw (1973), Mueller-Dombois & Ellenberg (1974) and Southwood ( 1978). Generally, the size of the sample unit used for diversity assessments should be chosen according to the nature of organisms under investigation, but a larger number of small units is usually more desirable than a smaller number of large units. Weed impact studies (Table 2) have utilised a range of sample unit sizes ranging from 0.01 $m^2$ in grassland; Tyser 1992) to a large 1250$m^2$ in temperate wetland (Anderson 1995) with a mean size of sample unit of 86m2. The " nested-plot technique" (Mueller-Dombois & Ellenberg 1974, p. 47) is a useful method for determining the minimal area required to capture a representative sample of species within a particular habitat and should be utilised more frequently by weed ecologists.

The number of samples or replicates needed to detect differences in populations is always an important consideration in the design of any experiment. Weed impact studies utilising the multi-site comparison approach generally have used relatively large sample sizes ranging from 9 (McCarthy 1997) to 594 (Griffin *et al.* 1989) with a mean sample size of 83 replicates. Where exploratory data are available, a method to determine the number of replicates needed to detect a given "true" difference between means is provided by Sokal & Rohlf (1969), but can and should also be determined by `power analysis' (Green 1989). The adoption of more rigorous procedures for assessing the adequacy of sampling is recommended as a standardised technique for assessing the impact of environmental weeds.

Pseudoreplication often occurs in weed impact studies, but should be rigorously avoided if sound conclusions are to be made from data. In mensurative experiments, e.g. multi-site comparisons, pseudoreplication is often the consequence of the actual physical space over which samples (e.g. Anderson 1995) are taken or measurements made being smaller or more restricted than the inference space implicit in the hypothesis being tested ( Hurlbert 1984). In manipulative experiments, e.g. removal or addition techniques, pseudoreplication will most commonly occur from the use of inferential statistics to test for treatment effects with data from experiments where treatments are not replicated (though the samples may be) or replicates are not statistically independent (Hurlbert 1984). The use of fire in weed experimentation, a management tool that is difficult to apply and contain, often involves pseudoreplicated sampling.

The basic principles of sampling design and statistical analysis relevant to the assessment and management of plant invasions are presented by Green (1979) and summarised well by Morrison (1997). These principles form the basis of scientific rigour and should be adopted as a standardised approach for weed impact studies.

### Data analysis

The conclusions drawn from weed impact studies are based on data analyses which range from simple descriptive statistics, e.g. Hobbs & Mooney (1986), to powerful and often sophisticated techniques such as the modeling techniques used by McIntyre & Lavorel (1994) to determine the impact of land-use and weed invasion on species diversity. Making recommendations on standardised analyses to determine the impact of weeds on biodiversity values seems of little value, as the analytical techniques adopted will largely be governed by the type and nature of the data collected and by the experimental design. In some cases the treatment of the data cannot be fully resolved until they have been collected and subjected to examination. It is possible, however, to identify techniques that have provided valuable insights into weed invasion impact or processes and these can be suggested for consideration in future studies.

The use of correlation statistics has been used effectively for many weed impact studies (Table 2). Measured variables relevant to a weed population (density, biomass, percentage canopy cover, etc.) can be correlated with other environmental or biological characteristics (species richness, overstorey density etc.) to produce a matrix of statistical relationships. Such correlations provide considerable information in an informative and succinct manner. Fensham *et al.* (1994) made excellent use of constellation diagrams to display a large range of Spearman Rank Correlation coefficients that enabled the complexity of a living system to be readily appreciated. As discussed earlier, although correlations can be informative and interesting, they are not conclusive and wherever possible, supplementary data or analyses should be included to either support or reject the original hypothesis. Use of ANOVA is often useful in this respect, especially in regard to manipulative experimental designs. Provided the underlying assumptions for ANOVA can be satisfied ( random allocation of treatments or samples, errors are normally distributed, variances are homogeneous, errors are independent), quantitative data from weedinvaded and weed-free sites can be compared statistically. Both parametric and non-parametric techniques have been utilised, depending on the nature of the data being analysed. Where the assumptions of ANOVA can be fulfilled, this technique provides a more accurate method of analysis than non-parametric methods. However, where the data cannot be assumed to be normally distributed, non-parametric methods such as the Mann-Whitney U-Test, KruskallWallis Test, Spearman's Rank Correlation Coefficient or the Wilcoxon Signed-Rank Test provide useful methods of analysis and have been used frequently in the interpretation of weed impact data.

**Figure 7.** **Sweet Pittosporum *(Pittosporum undulatum)* in moist sclerophyll forest, Mornington Peninsula, Victoria. Sweet Pittosporum is a tree native to eastern Australia, but has invaded habitats outside its native range. Dense infestations cause a substantial reduction in light intensity at the forest floor that leads to a reduction in the diversity of plants and animals. High levels of fire-retarding fuel reduces fire intensity and frequency which, in turn, can interfere with the regeneration of native plant species. Photo: R. Adair**

Mullett & Simmons (1995) made good use of pattern analysis to demonstrate the impact of *Pittosporum undulatum* on sclerophyll forest. A classification dendrogram was used to group quadrats into assemblages based on their inherent similarities. The assemblages, which showed close relationships between cover and abundance of *P undulatum,* were then used in a two-way table to illustrate the impact of the weed on species richness. Although rather "bulky", the two-way table demonstrates impact trends and provides data at a species-specific level. A "scatterplot" presentation of weed cover against species richness (e.g. Braithwaite *et al.* 1989) illustrates similar data in an abbreviated and more "generic" manner.

## A recommended approach for measuring impact of environmental weeds

It is important from the outset to begin with the right questions concerning the determination of the impact of environmental weeds on biodiversity values. In most cases, four important questions need to be answered, each of which will influence the specific techniques utilised to quantify weed impact:

1. What is the impact on plant and animal richness and diversity?

2. Is the impact affecting ecosystem-level functions, and if so how?

3. What interactions occur between vegetation succession and invasion?

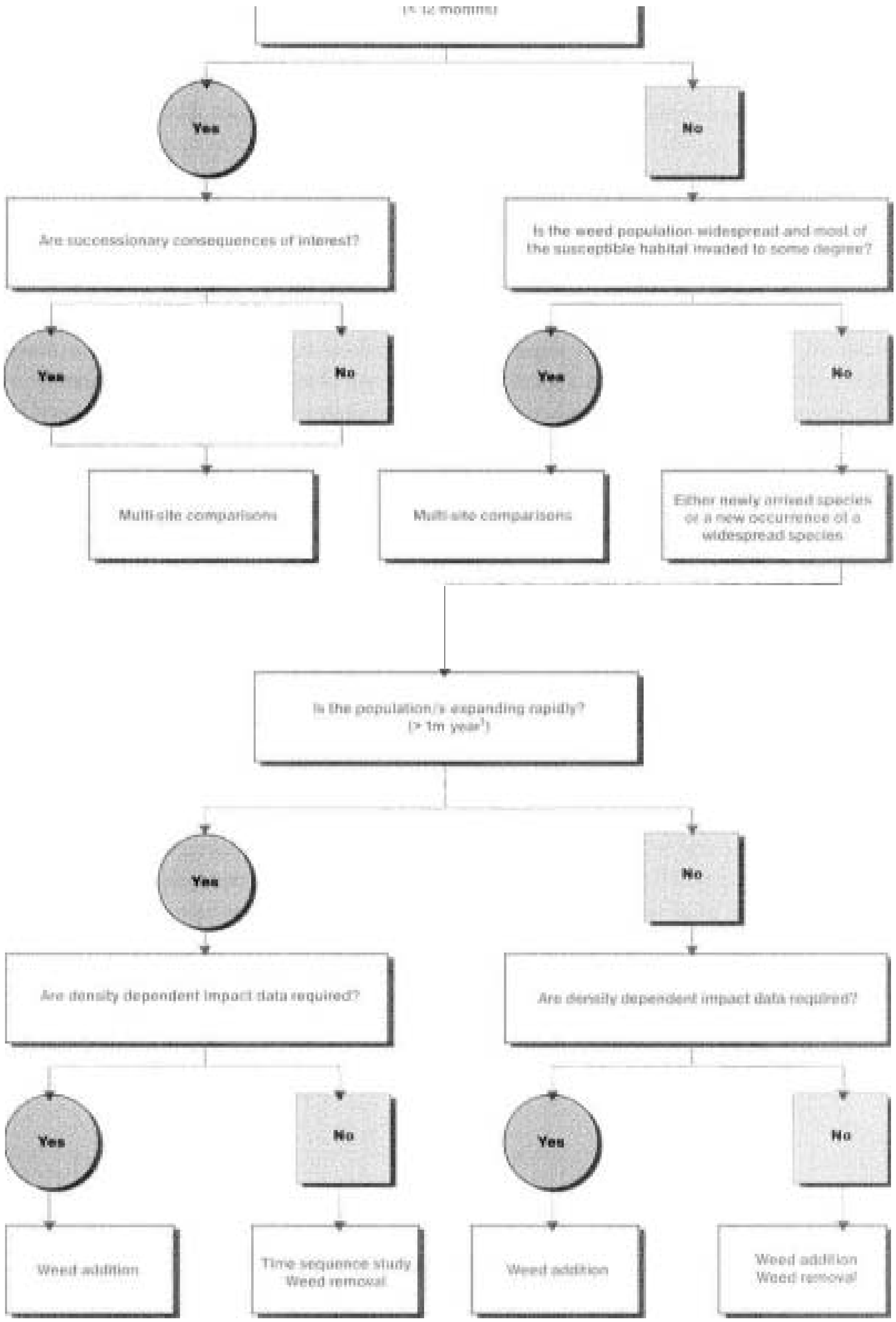
4. What are the threshold points for declines in biodiversity as a result of invasion?

**Figure 8.**

**A low infestation of *Chrysanthemoides monilifera* (Boneseed) at Arthurs Seat, Victoria. Low infestations of most weeds are likely to have no significant ecological impact. Determining the threshold of ecological impact provides a maximum tolerance level for infestations and a target for weed reduction programs. Photo: R. Adair**

The practicalities of ecological research, which are largely influenced by budgetary constraints, time allocation and project duration, will determine which and to what extent these questions can be answered. Whether all the questions can be addressed in one study is not critical, so long as the most relevant issues are addressed in relation to the protection or management of natural areas. While each of these questions are important and worthy of investigation, determination of the impact of weeds on ecosystem functions is perhaps the most critical. Weeds that cause disruption to essential ecosystem processes will inevitably cause serious declines in biodiversiry. Once the processes of impact are determined, then the critical questions should be directed at firstly, determining what are the stimuli that facilitate invasion and secondly, what level of population reduction is required to protect or enhance biodiversity values.

While there may be interest in developing a standardised model for assessing the impact of invasive plants, such an approach may hinder progress in developing an understanding of the ecology and impact of invasive species. Restricting research methodology by standardisation could unduly restrict impact assessment opportunities and tend to limit the type of data collected. It is more important for impact assessment procedures to be undertaken in a manner that focuses on the key issues in a sound quantitative and objective manner. Issues relating to the design of experiments, data collection processes and analyses have already been discussed in this report. All of the methods discussed so far (multi-site comparisons, weed removal, weed addition and time sequence studies) have relevance in the assessment of weed impact on biodiversity and there is no one technique that is superior to the others. The choice of assessment technique employed to evaluate the weed invasion impact will depend on the hypotheses and questions proposed at the commencement of the evaluation (see above), the stage of the invasion process and the time and resources available for the assessment. Providing that scientific rigour is applied, any one of the four methods is appropriate for evaluating weed impact. However, there are instances where one method may be more appropriate than another ( Figure 9). While all impact studies have utilised one of the methods described above, none has combined methods to evaluate weed impact. There may be reasonable arguments based on resource availability why such an approach cannot be adopted, but utilising more than one method to determine the impact of an invasive species on biodiversity values is by far the most powerful approach available. For example, combining weed addition and weed removal methods, which could be run synchronously, would identify which species are most susceptible to invasion effects as well as those that are most resilient. Combining the multi-site comparison approach with a weed addition experiment would provide much stronger proof that weed impact detected in the former approach was due to the direct effects of invasion and not to historical events that may have rendered habitats susceptible to invasion.



Yes
No
Are successionary consequences of interest?
Is the weed population widespread and most of the susceptible habitat invaded to some degree?
Yes
No
Yes
No
Multi-site comparisons
Multi-site comparisons
Either newly arrived species or a new occurrence of a widespread species
Is the population/s expanding rapidly? (> 1m year-1)
Yes
No
Are density dependent impact data required?
Are density dependent impact data required?
Yes
No
Yes
No
Weed addition
Time sequence study Weed removal
Weed addition
Weed addition Weed removal


**Figure 9.**

## Application of the multi-site comparison approach: two hypothetical examples

*Pittosporum undulatum* is an Australian native tree that is invading sclerophyll forests in southern Australia; *Brachiaria mutica is* an exotic grass that is invading tropical wetlands (Humphries *et al. 1991).* Impact assessment evaluation of *P undulatum* (Mullett & Simmons *1995)* has been undertaken but not necessarily following the techniques outlined above. The rationale behind technique selection for each weed is mentioned, indicating a process of application.

| | ***Pittosporum undulatum*** | ***Brachiaria mutica*** |
|---|---|---|
| Approach | Multi-site comparison. Easily applied as mosaic infestations are common in s.e. Australia. Control sites can be located between *Pittosporum* canopies (Mullet & Simmons 1995). | Multi-site comparison. Readily applied as mosaic infestations are common. Comparable water bodies with various levels of infestation can be compared. Time sequence studies would also be suitable due to rapid rate of spread. |
| Study site location/s | Wet or dry sclerophyll forest | Tropical freshwater wetlands |
| Sample unit location | Line transects through infestations (Mullett & Simmons 1995). | Line transects. Stratify sampling by establishing transects at intervals from shore to determine impact at range of water depths. |
| Sample unit size | Use Nested Plot Technique | Use Nested Plot Technique |
| Sample size ( replicate number) | Use method proposed by Sokal & Rohlf (1969). | Use method proposed by Sokal & Rohlf (1969). |
| Data collection | *Pittosporum* is slowly becoming accepted as a damaging environmental weed. Data on its impact on alpha diversity would build a more general appreciation of its impact in natural ecosystems. Many questions remain to be answered regarding its impact on fauna. Test alternative hypothesis that *Pittosporum* invasion occurs in species-poor forest gaps. | *Brachiaria* is rapidly spreading through tropical wetlands. No data are available on its impact. Data collection for determining biological impact should also collect information on environmental conditions that may elucidate invasion mechanisms. As wetlands are critical wildlife habitat, emphasis should be focused on impact on fauna. |
| Alpha diversity | As for Mullett & Simmons (1995). Measuring the biomass of understorey components would provide further evidence of impact. Data on impact on overstorey regeneration are also needed. | Determine species richness, percentage cover, biomass, stature of overstorey component. Collect data on water depth, disturbance history, nutrient status of water and stratum, flooding history, water availability. |
| Ecosystem-level changes | Determine number of regenerating overstorey species, N, P & K levels, fuel characteristics, light reduction levels, litter accumulation & decomposition rates, level of allelopathic compounds beneath *Pittosporum* canopies. | Determine impact of *Brachiaria* on water quality, particularly pH, nutrient levels, $0_2$, $CO_2$ levels. Measure light reduction and transpiration rates. Determine impact on hydrological cycles in wetlands. Measure litter accumulation levels. |

| | | |
|---|---|---|
| Gamma diversity | Determine habitats invaded, potential range of invader, distribution of susceptible habitats, distribution of "invasion triggers". | Overlaying data could enable impact assessment on a regional scale. |
| Succession | Age trees by basal stem diameters. Sample S & %CC in different age-classes. Count dominant tree & shrub seedlings in each stand. | Sample by aging invasion into wetlands from aerial photographs or from landholder knowledge. Sample S & %CC in different age-classes. |
| Fauna | *Pittosporum* creates heavy shade in the understorey and more mesic conditions than in uninvaded sites. Heat or sun-seeking-avoiding fauna are likely to be affected. Determine S & N for birds, reptiles, mammals, soil arthropods and forest invertebrates. | Tropical wetlands are critical habitat for a broad range of fauna. Sampling for changes in nesting and feeding patterns of wetland birds, species diversity of birds, amphibians, aquatic invertebrates and reptiles is needed. Sampling techniques for each group would be developed and applied to wetlands in various stages of invasion *by Brachiaria.* Stratification of sampling may be needed to account for differences in wetland type (shallow vs deep, flowing *vs* stagnant). |
| Analysis | As for Mullett & Simmons (1995). Develop correlation matrices, consider constellation diagrams for data collected from ecosystem-level studies. Model impact on fire behaviour using methods of van Wilgen & Richardson (1985). Plot rank abundance graphs. Calculate alpha log series index, Margalef & Berg-Parker indices. Test fit of main species model. | Consider pattern analysis with two-way tables to examine impact in different wetland types. Develop correlation matrices for invasion states and environmental parameters to elucidate invasion mechanisms. Establish constellation diagrams, plot rank abundance graphs. Calculate alpha log series index, Margalef & Berg-Parker indices. Test fit of main species model. |
| Publication | In either PPQ, Aust. J. Bot., Aust. J. Ecol., J. Appl. Ecol., Biol. Conserv., Oecologia. | In either PPQ, Aust. J. Bet. , Aust. J. Ecol., J. Appl. Ecol., Biol. Conserv., Oecologia, Trop. Grasslands. |

## Impact assessment - a view on priorities

As assessments of weed impact on biodiversity in Australian natural ecosystems are relatively rare, it is tempting to believe that any work directed to this field would be worthwhile and important. Is this the case? Certainly, widespread invasions have attracted the attention of weed ecologists and the few quantitative studies available report on the impact of important weeds such as *Lantana* (Fensham *et al. 1994), Mimosa* (Braithwaite *et al. 1989), Chrysanthemoides* (Weiss & Noble *1984)* and *Cytisus* (Smith *1994).* It is surprising, therefore, that longestablished, highly aggressive and rapidly spreading weeds such as *Cryptostegia grandiflora* (Rubber Vine), *Asparagus asparagoides* (Bridal Creeper), *Rubus discolor* (Blackberry) and *Acacia nilotica* (Prickly Acacia), to nominate just a few, have received virtually no attention in terms of their impact on biodiversity. Whilst it would be of great interest to quantify the impact of between the population parameter and biodiversity measures. This model predicts that any level of weed control will result in improvements in biodiversity values. Decisions on control targets will then largely be based on subjective judgments of what is an acceptable improvement or what resources are available to undertake the suppression program. The Type III model is a positive relationship between the weed and biodiversity parameters. This latter situation may occur where a weed supports critical processes or habitats required by native biota. Lastly, the previous three models assume either a positive or negative relationship between biodiversity and weed attributes, but there may be instances where there are no detrimental or beneficial impacts associated with invasion. This is the Type IV model and consists of a vertical linear line ( Figure 10). The methodology to determine threshold levels awaits development, but it may entail either repeated sampling using the multi-site comparison technique with a large replication level or, perhaps more powerfully, the adoption of the weed addition technique where the invader is planted and maintained at different densities in a range of vegetation communities. Monitoring impact at and following critical community events, e.g. regeneration after fire or other forms of disturbance, may provide useful insights into the mechanisms of weed impact.

Determination of management barriers to prevent or reduce the establishment of environmental weeds was a concept discussed by Hobbs & Hopkins (1990) and determined experimentally by Reader & Bricker (1994). The methods adopted by Reader & Bricker ( 1994) where "environmental triggers" to invasion were determined for exotic forbs in deciduous forests in North America, could be adopted for some management practices ( grazing, forestry, nutrient input) in Australia and are certainly worth considering further. The manipulative approach where controlled experimental treatments are applied to native vegetation would undoubtedly be the most useful technique in this regard. Hobbs & Atkins ( 1988) used a similar method to examine the relative impact of disturbance and nutrient addition to native vegetation in Western Australia but, although differences between treatments and vegetation types were identified, threshold levels were not determined.

New weeds are establishing frequently in Australian natural ecosystems, a trend that is likely to continue well into the future. Some new weeds appear to have considerable potential to degrade natural ecosystems but, as little is known about their ecology, control, or effect on biodiversity values, many escape attention. It is perhaps worthwhile to focus attention on determining the impact these weeds may have on biodiversity values, particularly alpha diversity levels, as a means of determining the potential threat these weeds may pose and to provide a factual base for generating greater awareness - both politically and socially - of these weeds within the Australian community. There are numerous weeds that fall into this category but *Annona glabra* (Pond Apple), an invader of tropical wetlands, *Polygala myrtifolia* (Myrtleleaf Milk Wort), a weed of coastal habitats in southern Australia, *Sollya heterophylla* (Western Australian Blue-bell Creeper), indigenous to south-west Western Australia but rapidly invading sclerophyll forests in eastern Australia, and *Undaria pinnatifida* (Japanese Kelp), an invasive alga in southern coastal waters, are all good examples of weeds that are bound to become more abundant in the future and could further degrade natural ecosystems. Grasses in Australian natural ecosystems have received scant attention in terms of their ecological impact, although several highly invasive species in northern Australia (viz. *Cenchrus ciliaris, Brachiaria mutica, Echinochloa polystachya, Hymenachne amplexicaulis, Pennisetum polystachion)* have received recent consideration ( Humphries *et al.* 1991). D'Antonio & Vitousek (1992), in their review of the ecological status of invasive grasses, clearly demonstrate the magnitude of the impact that this group of grass species may have on natural ecosystems. These range from alterations to diversity levels to substantial impacts on a range of ecosystem-level functions including fire intensity and frequency, level and rate of resource availability (particularly of nutrients), microclimate, geomorphological processes and successional pathways. It is highly probable that grasses have a profound effect on a broad range of plant and animal communities in Australia, both at the habitat and landscape level. Future research into elucidating their impact and the processes by which they degrade natural ecosystems are an obvious priority and would be a substantial contribution to weed science and weed management in this country and elsewhere.

The development of invasion models, where the impact of functional types on a particular habitat or landscape and, importantly, the identification processes that facilitate invasion, require greater attention from weed ecologists. Understanding invasion processes can allow resources to be better directed at controlling the source of invasion, which is usually inappropriate land management. Greater effort at analysing weed invasions from this perspective may yield stronger predictive and preventative capabilities than we have at present. It is also likely to result in greater weed control success and is applicable to multi-species control rather the "traditional" mono-species control approach. This conceptual approach is attracting the attention of modern weed scientists and Hobbs & Humphries (1995), Panetta & Lane (1996), Woods (1997) and Luken (1997) all draw attention to its ecological value. Once an understanding of invasion processes are available, management of the invasion may then commence. The approach proposed by Luken (1997) where succession management involving designed disturbance, controlled colonisation, and controlled species performance may then replace traditional `weed control' practices.

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# Appendix

Victorian species or communities listed under the Victorian *Flora and Fauna Guarantee Act* 1988 with Action Statements where environmental weeds are considered a potential threat

| ***Species*** | ***Weed*** | ***Impact*** |
|---|---|---|
| ***Plants*** | | |
| *Acacia maidenii* (Maiden's Wattle) | *Rubus fruticosus, Erica lusitanica* | Competes with seedlings |
| *Adiantum capillus-veneris* (Dainty Maidenhair) | *Coprosma robusta, Hedera helix* | *Coprosma a* potential competitor, but may offer some shade. *Hedera* smothers plants. Weeds may limit dispersal of *Adiantum* |
| *Adiantum diaphanum* (Filmy Maidenhair) | N/A | Habitat invasion |
| *Astelia australiana* (Tall Astelia) | *Rubus laciniatus* | Competition |
| *Caladenia audasii* (Audas' Spider Orchid) | N/A | N/A |
| *Caladenia calcicola* (Limestone Spider Orchid) | *Acacia sophorae* | Smothers habitat |
| *Discaria pubescens* (Hairy Anchor Plant) | *Cytisus scoparius, Ulex europaeus, Rubus fruticosus* | N/A |
| *Diuris cuneata* (Wedge Diuris) | *Chondrilla juncea, Echium plantagineum, Juncus acutus, Hypericum perforatum, Romulea rosea, Paspalurn, Cirsium, Avena, Bromus, Hypochoeris* | N/A |
| *Diuris fagrantissima* (Sunshine Diuris) | *Stipa neesiana, Foeniculum vulgare, Romulea rosea* | N/A |
| *Eucalyptus crenulata* (Buxton Gum) | *Rubus futicosus, Lonicera japonica* | N/A |
| *Eucalyptus fogattii* (Kamarooka Mallee) | N/A | Competition with seedlings |
| *Euphrasia sea bra* (Rough Eyebright) | *Rubus f uticosus* | Competition, particularly on seasonally inundated ground |
| *Psoralea parva* (Small Psoralea) | *Echium plantagineum, Bromus diandrus* | N/A |
| *Rutidosis leptorhynchoides* (Button Wrinldewort) | *Stipa neesiana, Foenieulum vulgare* | N/A |
| *Senecio macrocarpus* (Large-fruit Groundsel) | Annual grasses | Competition |
| *Westringia crassifolia* (Whipstick Westringia) | *Briza maxima* | Reduces natural regeneration |

| | | |
|---|---|---|
| **Animals** | | |
| *Burramys parvus* (Mountain *Pygmy* Possum) | N/A | N/A |
| *Delmar impar* (Striped Legless Lizard) | *Stipa neesiana* | Habitat degradation |
| *Lichenostromus melanops cassidix* (Helmeted Honey Eater) | N/A | N/A |
| *Paralucia pyrodiscus lucida* (Eltham Copper Butterfly) | *Genista monspessulana, Pinus radiata, Briza maxima, Rubus fruticosus, Ehrharta calycina* | Habitat degradation |
| *Pomatostomus temporalis* (Grey-crowned Babbler) | *Phalaris,* pasture grasses | Reduces foraging habitat |
| *Tympanocryptis lineata pinguicolla* (Southern Lined Earless Dragon) | N/A | Degrades habitat |
| **Communities** | | |
| Butterfly Community No. 1 | Thistles, *Senecio pterophorns* | Alters vegetation structure & composition |

# Glossary of terms

| | |
|---|---|
| Species richness (S) | Number of species within a defined area |
| Species diversity | A combination of the number of species present and the way in which the individuals are distributed amongst the various species |
| Species density (N) | Number of individuals of a particular species within a defined area |
| Species evenness (E)<br>(Pielou 1969) | A measure of the evenness of species' abundance<br>$E = \frac{H'}{H'max.} = \frac{H'}{\text{log total no. species}}$ |
| Species frequency (F) | Number of sampling units that a species occurs in |
| Canopy cover (%CC) | The percent ground cover of a species' foliage canopy |
| Biomass (B) | The weight of material present within the study unit |
| Alpha diversity | Species diversity within a specified habitat |
| Beta diversity | Species diversity between habitats |
| Gamma diversity | Species diversity within a specific geographic area |
| Shannon-Weiner Index | $H' = -Y\text{-}p_i \ In\ p_i$<br>where $p_i$, the proportional abundance of the ith species= (ni/N)<br>e.g. $p_i$ = av absolute density of species $p_i$<br>l,av. absolute density of all species |
| Margalef Index | Ding = (S-1)/In N<br>where S=number of species, N = number of individuals |
| Berger-Parker Index | D = Nmax/N<br>where N=total number of individuals, Nmax = number of individuals in the most abundant species. The reciprocal form is usually adopted, i.e. 1/D |
| Alpha Log Series Index | Alpha= N(1-x)/x<br>where x = an iteration of S/N=[(1-x)/x] [-In(1-x)] with S= total number of species, N= total number of individuals. See Magurran (1988) p. 132, for a worked example |
| Genet | A non-clonal individual; usually the result of sexual reproduction |
| Ratnet | An individual member of a clone |