Does environmental weed control achieve conservation gain?

A literature review

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Abstract

Millions of dollars are spent controlling weeds in natural ecosystems all over the world, but does the control necessarily result in conservation gain, such as an improvement in native plant recruitment or an increase in the native invertebrate population? The aim of this review was to locate and summarise studies that investigate this question, and attempt to identify any common cause behind the success or failure of weed control programmes. We report on 51 weed removal experiments from 48 studies (15 from New Zealand), assessing the effects of weed control on some aspect of native biodiversity. Thirty-eight experiments compared the effects of one or more treatment methods with untreated areas where the weed population remained intact. Thirteen of the experiments measured the effects of one or more treatment methods, but did not compare results with untreated areas. Almost two-thirds of the experiments had at least one treatment that resulted in positive conservation gain. However, only nine of the experiments reported positive conservation gains for all treatments. Thirty-five percent of the experiments included at least one treatment that failed to achieve any positive conservation gain. The reasons for failure were not always apparent, but regeneration of the target weed, invasion by other weeds and native seed limitation appear to be common factors limiting success. We also looked for studies that investigated whether different levels of weed control resulted in different outcomes for native biodiversity, but found very few. Many studies report a negative linear relationship between weed abundance and native species richness and/or cover. However, statistical methods used were often inadequate, and a non-linear threshold relationship was apparent from some studies, so it is uncertain how much weed populations need to be reduced by in order to achieve conservation gain. In summary, the studies we reviewed were highly variable in terms of the species and ecosystems studied, experimental designs and statistical methods used, and outcomes achieved. Accordingly, we were unable to identify universal factors that determine whether weed control programmes achieve conservation gain.

Keywords: environmental weeds, invasive species, weed control, conservation, biodiversity

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1. Introduction

Weeds can have detrimental effects on many different aspects of conservation concern. For example, they can reduce the abundance and diversity of native flora and fauna (Braithwaite et al. 1989; Maloney et al. 1999; Standish et al. 2001; Gaertner et al. 2009), they can adversely affect natural ecosystem processes such as hydrological and sedimentary regimes (Görgens & Van Wilgen 2004; Hilton et al. 2005), and they can alter natural disturbance regimes (Mack & D'Antonio 1998). Land managers commonly assume that controlling weeds will eliminate these detrimental effects, but management outcomes are rarely measured, so there are few data on which to base an assessment of the validity of this assumption (Harms & Hiebert 2006; Raymond et al. 2006). This was illustrated by a recent review of the management of Australia's 20 'Weeds of National Significance' (WONS) in natural ecosystems—only 18 out of 95 published reports included an assessment of the response of other plant species to the management of the weed (Reid et al. 2009). Additionally, the authors surveyed land managers involved in the control of WONS and found that while 86 of the 142 respondents did evaluate their WONS management programme, in most cases the evaluation was qualitative, rather than quantitative (Reid et al. 2009). Quantitative evaluation of the effects of weed control on native biodiversity (often called outcome monitoring) is an expensive, specialist job, so it is perhaps not surprising that it is rarely undertaken. Indeed, the primary goal of most weed control programmes is to eliminate the weed, regardless of what happens afterwards.

The aim of this literature review was to summarise studies that attempted to quantify the response/s of native biodiversity following weed control. In doing so, we hoped to identify any common factors that lead to conservation gain (or loss) following weed control. Conservation gain can be generally defined as any positive outcome for native biodiversity. This could include gains achieved by preventing weed invasion in the first place, but our focus is on what happens after management in weed-infested areas. The variable most commonly measured following weed control (other than whether the weed died) is some aspect of the native plant community, such as proportion of native plant cover, or native species diversity.

We used Google Scholar to search for published studies that included various combinations of the following terms—weed control, weed biomass, weed threshold, biomass threshold, environmental weed, vegetation recovery, environmental outcome monitoring, native regeneration, conservation outcome, native regeneration, conservation, and biodiversity. We also searched the Department of Conservation (DOC) library catalogue for DOC reports on these topics, and checked the contents pages of key journals from 2000 to 2009 (*Restoration Ecology, Journal of Applied Ecology, Ecological Applications, Conservation Biology, New Zealand Journal of Ecology* and *Plant Protection Quarterly*) for relevant articles, and found additional studies in the reference lists of previously located articles or theses. In particular, we looked for studies carried out in New Zealand, but we also included international studies.

The review covers the following general topics:

- Section 2 is the main summary of studies that have measured the effects of weed control on native biodiversity, and is grouped according to the research approach taken
- Section 3 reviews studies that considered whether different levels of control resulted in different levels of conservation gain
- Section 4 is a discussion of weed impacts and how they can be measured, and includes the concept of weed control thresholds—the critical point at which weed control should be undertaken in order to avoid conservation loss
- Section 5 considers the effects of post-control management on native biodiversity

This review is the initial stage in a research project that aims to optimise environmental weed management—reducing costs while still achieving conservation gain.

2. Effects of weed control on native biodiversity

Adair & Groves (1998) identified three main approaches to determining weed impacts on biodiversity: multi-site comparison, weed removal, and weed addition. Although only the weed removal approach directly measures the effects of weed control, both multi-site comparisons and weed addition experiments can provide useful insights into the effects of weeds and variations in their abundance on biodiversity, and from this the effects of weed control can be inferred.

2.1 Multi-site comparisons

The multi-site comparison approach measures selected biodiversity parameters at sites where the weed is present and compares them with sites where the weed is absent. The assumption being made with this approach is that any differences between invaded and uninvaded sites are likely to be caused by the presence or otherwise of the weed. However, no manipulative experiments are conducted using this approach, and causation of effects cannot usually be demonstrated (Brown & Sax 2004). For example, the vegetation composition may have been different in invaded areas prior to weed invasion (Adair & Groves 1998) or environmental factors may differ, influencing whether a site is invaded or not. Despite this limitation, this is a commonly used approach to infer weed impacts. Almost without exception, the studies we found that used the multi-site comparison approach all reported that the aspect of native plant biodiversity measured (abundance, diversity, species richness, etc.) was lower at sites where the weed was present. Several studies used a combination of multi-site comparison and weed removal experiments, and these are reported in section 2.2 below.

Jackson (2005) used the multi-site comparison approach to investigate the impacts of the invasive buffel grass (*Cenchrus ciliaris*) in north-eastern Queensland. Sites dominated by buffel grass had fewer herbaceous species and fewer native species than sites without buffel grass (Jackson 2005). The number of exotic species did not differ significantly between plots with and without buffel grass (Jackson 2005). A multi-site comparison was also used to measure the impact of bridal creeper (*Asparagus asparagoides*) on native plants at four sites in Western Australia (Turner et al. 2008). Native species richness and percentage cover was significantly lower in 'invaded' plots (bridal creeper cover 40.3–61.7%) than 'uninvaded' plots (0.2–5.1%) (Turner et al. 2008). The number and percentage cover of exotic species did not differ between bridal creeper-invaded plots and uninvaded plots (Turner et al. 2008). Across all four sites, 31 native species were found only in uninvaded plots, while six native species were found only in bridal creeper plots (Turner et al. 2008). A similar study conducted in South Australia found that species richness and frequency of native ground cover vegetation was lower in bridal creeper-invaded than uninvaded plots (Stephens et al. 2008).

At two Northland sites in New Zealand, native woody seedling density and species richness were significantly higher in areas where wild ginger (*Hedychium gardnerianum*) was sparse than in areas where wild ginger was dense; this was true even when the native seed rain was greater in dense wild ginger plots (Williams et al. 2003). Across sparse wild ginger plots, small-seeded species were consistently abundant in both the seed rain and as seedlings (Williams et al. 2003). In contrast, there were few seedlings of small-seeded species in dense wild ginger plots, even though they were still abundant in the seed rain (Williams et al. 2003). Large-seeded natives (kohekohe (*Dysoxylum spectabile*), karaka (*Corynocarpus laevigatus*) and supplejack/kareao (*Ripogonum scandens*)) were the most abundant seedlings in dense wild ginger plots, despite being uncommon in the seed rain (Williams et al. 2003). The dense cover of invasive forest floor weeds appears to favour the establishment of large-seeded species (Standish et al. 2001; Williams et al. 2003). Seed size is often positively correlated with shade tolerance, and large seed size can therefore be advantageous for seedling survival in deep shade (Fenner & Thompson 2005).

Ogle et al. (2000) investigated the effects of old man's beard (OMB) (*Clematis vitalba*) and its sporadic control over several decades on native biodiversity in a forest remnant in Manawatu, New Zealand. Native tree and shrub species richness and frequency was higher where OMB was absent (15 species) than where it was present (11 species, eight of which had only one individual). Native shrub density was more than eight times higher in plots without OMB than in plots with OMB (Ogle et al. 2000). Native shrubs were also shorter in areas with OMB, probably due to sheep grazing that was used as part of control efforts. Of the 146 species recorded in the forest remnant between 1946 and 1972, only 111 (76%) have been recorded since 1990 (Ogle et al. 2000). Herbaceous flowering plants were most severely affected, with 37% of species lost. Shrubs and small trees lost 24% of species, and there were no losses of tall trees (Ogle et al. 2000). Nationally rare plants were disproportionately affected by OMB, losing 44% of species (Ogle et al. 2000).

In braided river habitat in the South Island, only three out of 327 river bird nests monitored at three sites were found in crack willow (*Salix fragilis*) habitat, even though willow and non-willow habitats were sampled equally (Maloney et al. 1999). Two of the four river bird species monitored had lowest densities at the site where willow infestation was greatest (Maloney et al. 1999).

The effect of bone-seed (*Chrysanthemoides monilifera* ssp. *monilifera*) on native vegetation was investigated by comparing invaded and uninvaded plots in a eucalyptus woodland in Australia (Thomas et al. 2005). Many native species were more abundant in uninvaded plots than invaded plots, while diversity was lower in invaded plots (Thomas et al. 2005). In acacia-infested areas of South African fynbos, native plant densities were significantly lower and extinction probabilities were greater than in adjacent acacia-free habitat (Musil 1993).

Mason & French (2007) used a variation of the multi-site comparison to investigate the impacts of bitou bush (*Chrysanthemoides monilifera* ssp. *rotundata*) by including sites where the weed had already been controlled for conservation purposes, as well as invaded and uninvaded (mean bitou bush cover of 5%) sites. Bitou bush invasion appeared to have adversely affected coastal communities, with higher native species richness in non-invaded sites than invaded sites. Bitou bush control did not fully restore sites, as managed sites had significantly fewer native species than uninvaded sites and bitou bush still occurred at all managed sites at abundances ranging from 3.3 to 16.6% cover. Intensively managed sites had greater native species richness than extensively managed sites, but neither management category differed from unmanaged sites, which had an intermediate number of native species (Mason & French 2007).

Harms & Hiebert (2006) used a similar approach to investigate the effects of cutting or burning to control tamarisk (*Tamarix* spp.) at 29 sites in the USA where treatment had occurred between 1 and 11 years prior to the study. At each site, one transect was sampled in each of three treatment areas: cut-stump, burned and untreated controls. Mean tamarisk cover was 82–95% lower in both types of removal transects than in untreated areas. Native vegetation cover was significantly greater in treated than untreated areas, but community composition was not significantly different. Native species richness was nearly twice as high on treated transects in one of the three regions in the study, although untreated transects had 1.5 times more species in another region. Overall diversity and evenness was greater in treated than untreated areas (Harms & Hiebert 2006).

Two recent meta-analyses have confirmed that invasive plant species can have significant detrimental effects on native biodiversity (Gaertner et al. 2009; Mason et al. 2009). Mason et al. (2009) analysed 20 studies and found that both graminoid (grass and grass-like species) and woody (shrub and tree species) invaders had similar negative effects on native plant species richness. However, negative effects were greater for native graminoids than trees, perennial than short-lived species, and small-seeded than large-seeded species. Another recent meta-analysis of 47 studies confirmed that significant declines in native plant species richness in Mediterranean-type ecosystems could be attributed to invasive plants (Gaertner et al. 2009). These analyses showed that studies conducted at smaller scales or over longer time periods revealed stronger impacts of alien invasion than studies conducted at larger scales or over shorter time periods.

Gaertner et al. (2009) found that annual herbs, trees, and vines had the greatest impact, whereas graminoids generally caused insignificant changes for native plant communities. Additional references for comparative studies of the effects of weeds on native species are contained within both reviews (Gaertner et al. 2009; Mason et al. 2009).

In summary, most of the studies we found that used the multi-site comparison approach reported that the aspect of native plant biodiversity measured (abundance, diversity, species richness, etc.) was lower at sites where the weed was present. Furthermore, several recent meta-analyses have confirmed that invasive plants can have negative impacts on native plant populations. However, in general, these studies demonstrated correlation but not causality.

2.2 Weed removal studies

Both the weed removal and weed addition approaches fall into the category of neighbour manipulation experiments (Aarssen & Epp 1990). Neighbour manipulation experiments require long monitoring times to assess weed impacts, but have a range of advantages over comparative studies (Adair & Groves 1998). Where a weed species is abundant and widespread it can be difficult to find suitable study sites where the weed is absent or at low densities (Adair & Groves 1998), and there may be good ecological reasons why the weed is not there. In these situations, weed densities can be manipulated by removing the weed from discrete areas and monitoring the vegetation in comparison with adjacent weedy areas. In addition, manipulative experiments can demonstrate causation of weed and/or weed control effects. When weed removal experiments are conducted, non-invaded reference sites can be useful for determining the target for restoration, and the degree of success for control programmes (Turner & Virtue 2006).

Many studies have used weed removal experiments to determine the effects of weed control on native biodiversity compared with adjacent untreated plots (see below and Appendix 1 for a summary). Aarssen & Epp (1990) identified a number of potential limitations of removal experiments, including soil disturbance during removal, soil compaction during monitoring and unknown effects of leaving the root material in the soil. However, where the aim is to determine the effect of removing the weed rather than measuring the impact of the weed, these limitations are less relevant. There are a large number of studies in this section, so they have been grouped according to area of origin—New Zealand or international. Additionally, the international studies have been grouped according to the life form of the weed species being controlled.

2.2.1 New Zealand weed removal studies

Several experimental studies quantifying the effect of weed removal on native biodiversity have been conducted in New Zealand (Williams et al. 1998; Husted-Andersen 2002; Standish 2002; Barton et al. 2007; Bourdôt & Saville 2008; Paul & Ledgard 2008a, 2009). Two biological control agents (white smut fungus, *Entyloma ageratinae* and a gall fly, *Procecidochares alani*) were introduced to northern New Zealand to suppress mist flower (*Ageratina riparia*), a serious environmental weed (Barton et al. 2007). Mist flower abundance declined significantly 5 years after the first biocontrol agent was introduced (Barton et al. 2007). Native species richness was initially significantly lower in mist flower-invaded plots than in paired control plots lacking mist flower, while similar differences in native species percentage cover approached statistical significance. As mist flower abundance declined, both native species richness and native species percentage cover in mist flower-present plots became increasingly similar to levels in plots where mist flower was absent (Barton et al. 2007). In contrast, there was no significant difference in exotic species richness or cover between plots in which mist flower was present and those where it was absent (Barton et al. 2007).

Williams et al. (1998) measured the response of native vegetation to removal of the weed species Japanese honeysuckle (*Lonicera japonica*), climbing dock (*Rumex sagittatus*) and bone-seed (*Chrysanthemoides monilifera* ssp. *monilifera*) in an unreplicated study in New Zealand. After 1 year, Japanese honeysuckle had resprouted and seedlings dominated areas where it had been cleared, along with other weed species (Williams et al. 1998). Native vegetation showed signs of recovery 1 year after dock removal, with 94% of dock plants killed (Williams et al. 1998). There was no native vegetation recovery following bone-seed control, possibly because of a dramatic increase in the cover of exotic grasses (Williams et al. 1998). McAlpine et al. (2009) also looked at the effect of bone-seed removal on seedling regeneration, and found that bone-seed seedling abundance increased dramatically following control, whereas native seedling abundance decreased dramatically.

At three Canterbury sites, native vegetation cover was highest in areas where wilding conifers were felled < 10 years ago, intermediate in areas that never had wildings and lowest in areas where conifers were felled > 10 years ago (Paul & Ledgard 2009). Native species richness was significantly higher in areas without wildings (7.2 species) compared with felled-conifer plots, regardless of the length of time since felling (2.4–2.9 species). At one Canterbury site, total vegetation cover was significantly lower in wilding-mulched areas compared with unmulched areas after 1 year (Paul & Ledgard 2009). In the Marlborough Sounds, mean native woody species richness was significantly higher after 9 years where pines (*Pinus radiata*) were stem-poisoned (29 species) than where they were felled (9 species) (Paul & Ledgard 2009). Felled plots were dominated by exotic species including regenerating pines, gorse (*Ulex europaeus*) and cutty grass (*Gahnia lacera*). A previous study in Canterbury compared vegetation in areas where *Pinus nigra* trees were felled with adjacent open grassland (Paul & Ledgard 2008a). Plant species richness and percentage cover of the invasive *Hieracium pilosella* were both lower in felled tree plots than in open grass plots, while grasses and native shrubs were taller (Paul & Ledgard 2008a). Whether these results were due to the effects of tree logs remaining in the plots and/or *P. nigra* trees having grown in the plots over a period of 12–15 years prior to felling (and not in the open grass plots) is uncertain.

The effects of controlling grey willow (*Salix cinerea*) using aerial spraying of glyphosate and cutting with stump application of picloram on non-target vegetation were investigated in the South Taupo Wetland (Husted-Andersen 2002). There were no positive effects of control on species diversity or abundance 1 year after treatment (Husted-Andersen 2002). An experimental control programme was also established in the summer of 2002/03 for grey willow and crack willow at Kopuatai peat dome on the Hauraki Plains, Waikato (Reeves & Champion 2003). Glyphosate was applied aerially to alternate (treated/untreated) 30–50 m strips of grey and crack willow (Reeves & Champion 2003). For native plants, there were no significant differences in species cover, total native vegetation cover, or native species richness between treated and untreated plots after 28 months (Champion 2006a). The invasive reed canary grass (*Phalaris arundinacea*) was thought to be inhibiting native regeneration in plots where willow was removed (Champion 2006a), and subsequent grass removal trials were conducted (Champion 2006b). Removal of reed canary grass using Gallant™ herbicide significantly increased native sedge, woody and aquatic species cover after 1 year compared with before removal (Champion 2006b). Maloney et al. (1999) found that willow removal appeared to increase foraging and nesting habitat for some braided river bird populations in the Mackenzie Basin.

Standish (2002) conducted removal experiments for tradescantia (*Tradescantia fluminensis*) to investigate the effect of herbicide, hand-weeding and season of treatment on native vegetation in a Manawatu forest remnant. There was no difference in native seedling abundance or native seedling species richness after 20 months, while native plant survival decreased in herbicide-treated plots compared with hand-weeded and untreated plots (Standish 2002). Clearing tradescantia had no effect on the survival of transplanted native saplings after 2.5 years (Standish

2002). In another study, removal of tradescantia also had no effect on invertebrate abundance compared with untreated plots after 7 weeks, although invertebrate taxa richness was greater in hand-weeded plots than herbicide-treated or untreated plots (Standish et al. 2002).

A trial using three herbicide types (2,4-DB, Versatill™ and glyphosate) to control *Plantago coronopus* in salt patch communities in Central Otago found both 2,4-DB and Versatill™ were similarly effective at killing the weed and glyphosate was least effective (Bourdôt & Saville 2008). An additional trial was conducted to determine the effect of 2,4-DB and Versatill™ on the abundance of native *Lepidium kirkii* plants. There was no significant difference in the density and frequency of *L. kirkii* between plots sprayed with 2,4-DB or Versatill™ or untreated plots (Bourdôt & Saville 2008).

2.2.2 International weed removal studies

Grass removal

Removal of two invasive grass species (*Poa pratensis* and *Dactylis glomerata*) from a dry oak savannah in Canada resulted in increased cover of several native forbs, decreased cover of exotic forbs and non-obligate (associated with mesic forests) native forbs, no change in nearly half the species (including many rare species), and a significant increase in cover of the exotic *Cytisus scoparius* (MacDougall & Turkington 2005). Weeding significantly increased light and bare soil but had no effect on soil moisture, organic matter, or NO_3-NH_4 (MacDougall & Turkington 2005). The percentage cover of the native grass *Eragrostis variabilis* increased in plots where the invasive grass *Cenchrus echinatus* was removed compared with untreated plots on Laysan Island, Hawai'i (Flint & Rehkemper 2002). Removal of *Cenchrus* is likely to have a positive, indirect effect on wildlife, as the number of active seabird nests increased with increasing *E. variabilis* percentage cover (Flint & Rehkemper 2002). Cox & Allen (2008) examined the effects of different grass control methods on native (and exotic) forb species in southern California. Precipitation levels influenced cover of both grasses and forbs during the period of the study, and cover of individual native forb species fluctuated greatly (Cox & Allen 2008).

Large-scale weeding of an invasive perennial bunchgrass (*Schizachyrium condensatum*) in Hawai'i increased the biomass of two of the four native shrub species measured 3 years after removal, while growth in a third species shifted from the upper trunk to the lower 40 cm (D'Antonio et al. 1998). In addition, seedlings of all but one native woody species were significantly more abundant where bunchgrass was removed (D'Antonio et al. 1998). Bunchgrass probably reduced growth of some native species in part by depleting available nitrogen, which is a limiting resource in the study system (D'Antonio et al. 1998). All but one native shrub species had higher leaf tissue nitrogen in removal plots than control plots and available soil nitrogen levels were also generally higher in removal plots (D'Antonio et al. 1998).

The invasive Japanese stiltgrass (*Microstegium vimineum*) was removed from USA forests using three treatments: hand-weeding, post-emergent herbicide, and post-emergent herbicide followed by pre-emergent herbicide (Flory & Clay 2009; Flory 2010). After 2 years, hand-weeded plots were significantly reinvaded, while the two herbicide treatments resulted in very low stiltgrass cover (Flory 2010). Hand-weeding and post-emergent herbicide resulted in greater native plant community biomass and cover than untreated plots, but there was no difference between pre-emergent plus post-emergent herbicide and untreated plots (Flory 2010). Native community diversity was 24% greater when Japanese stiltgrass was hand-weeded and 21% greater when it was removed by post-emergent herbicide, but no greater using post-emergent plus pre-emergent herbicide than in untreated plots (Flory & Clay 2009). Native grass species richness was greater with hand-weeding and lower with post-emergent plus pre-emergent herbicide compared with untreated plots (Flory & Clay 2009). Removal of stiltgrass with post-emergent herbicide decreased native graminoid biomass compared with untreated plots, but native forb biomass increased for all removal treatments (Flory & Clay 2009).

Simmons et al. (2007) tested the effects of removing king ranch bluestem (*Bothriochloa ischaemum*) on dominant species using five different methods at two grassland sites in Texas, USA. Mowing treatments had no effect on the abundance of bluestem or dominant native and exotic species. Fire treatments reduced bluestem cover, but had a positive effect on the native forb *Solidago canadensis* and no effect on the native grass *B. lagaroides* or on two other non-native grasses (*Dicanthium annulatum* and *D. sericeum*). Glyphosate herbicide significantly decreased *B. ischaemum* growth at the site that had a high application concentration and, to a lesser extent, with repeated treatment at the other site. There was no effect of glyphosate on any of the other species.

Reinhardt Adams & Galatowitsch (2006) controlled reed canary grass in wet meadows in the USA using a combination of burning and herbicide treatments. Burning did not reduce reed canary grass abundance but did reduce its seed bank, while herbicide application was more effective in late August and late September than in mid-May (Reinhardt Adams & Galatowitsch 2006). Two suboptimally timed mid-May herbicide applications provided control equivalent to a single appropriately timed late-season application (Reinhardt Adams & Galatowitsch 2006). Reed canary grass had significantly recolonised treated plots 1 year after native seed sowing (Reinhardt Adams & Galatowitsch 2006). Limited establishment of native species occurred both from sown seed and naturally, although repeated herbicide application enhanced native colonisation (Reinhardt Adams & Galatowitsch 2006).

Burning combined with herbicide application or hand-weeding of invasive buffel grass (*Cenchrus ciliaris*) in Hawai'i effectively restored native grass communities within 2–4 years of control (Daehler & Goergen 2005). Annual and twice-yearly mowing of *Molinia arundinacea*-invaded wetlands increased vascular plant and bryophyte species richness and evenness (indicating a decrease in the cover of dominant species and an increase in competitively weak species) compared with untreated plots (Hájková et al. 2009). Native species abundance increased 18 months after the invasive fountain grass (*Pennisetum setaceum*) was controlled using a range of methods in Hawaiian dry forest (Cabin et al. 2002). Previous research had demonstrated that little native recruitment occurred without fountain grass control (Cabin et al. 2000). Aerial herbicide application to control the invasive *Euphorbia esula* negatively affected a number of native forb species and a native subshrub in a Montana, USA, grassland (Rinella et al. 2009). Herbicide application appeared to actually increase the abundance of the target weed *E. esula*, possibly due to reduced competition from native species (Rinella et al. 2009).

Vine removal

Alvarez & Cushman (2002) used disturbance plots to control for the effects of soil and vegetation disturbance caused by weeding in three coastal habitats in northern California. In their removal experiment, Cape ivy (*Delairea odorata*, called German Ivy in New Zealand) reduction increased plant species richness, seedling abundance and diversity for both natives and non-natives, compared with control plots (Alvarez & Cushman 2002). In a complementary comparative study, plots invaded by Cape ivy had significantly lower plant species richness, species diversity and seedling abundance than uninvaded plots (Alvarez & Cushman 2002). These results were generally consistent across three different habitat types (Alvarez & Cushman 2002).

Biggerstaff & Beck (2007) showed that different methods of controlling English ivy (*Hedera helix*) had different effects on the subsequent native plant regeneration. Although spraying with herbicide was effective at removing the ivy, hand weeding resulted in the greatest density and diversity of seedlings (Biggerstaff & Beck 2007).

Removal of the invasive bridal creeper in South Australia had no effect on native species richness after 8 years, even though bridal creeper was controlled to low levels (0.1–2.4% cover; Turner & Virtue 2006). Bridal creeper removal also had no effect on the abundance of many native species, but increased the density, biomass and percentage cover of native saltbush, and increased the percentage cover of perennial native grasses and exotic *Oxalis pes-caprae* (Turner & Virtue 2006). A complementary study that assessed the effect of bridal creeper removal following fire also found no difference between treated and untreated plots in native species richness, native plant composition or cover of most plant groups after 10 years (Turner & Virtue 2009). However, native small shrubs and native creepers and climbers had slightly higher cover on untreated plots (Turner & Virtue 2009).

Forb removal

Removal of *Impatiens glandulifera* for 3 months in a Czech Republic river valley had no effect on species richness, diversity and composition (Hejda & Pyšek 2006). A complementary comparative approach found that evenness was higher in uninvaded plots than invaded plots, but found no other significant differences (Hejda & Pyšek 2006). In contrast, removal of *I. glandulifera* from riparian habitat in northeast England increased total species abundance, richness, diversity, and seedling density, but decreased evenness (Hulme & Bremner 2006). For native species, *I. glandulifera* removal increased species richness and diversity, but had no effect on evenness (Hulme & Bremner 2006). Species richness and diversity of non-natives also increased after *I. glandulifera* removal but their evenness declined (Hulme & Bremner 2006). Because non-native species responded more strongly than natives to *I. glandulifera* removal, there were proportionally more non-native species in treated plots (Hulme & Bremner 2006).

Stinson et al. (2007) also used both comparative and experimental approaches to determine the effect of garlic mustard (*Alliaria petiolata*), and its removal, on native vegetation in forested habitat in New England, USA. The comparative approach revealed that native species diversity and tree seedling abundance declined with increasing garlic mustard abundance, but there was no correlation with native species richness or presence of non-native species (Stinson et al. 2007). The experimental approach had three treatments: complete removal (all garlic mustard hand-weeded), partial removal (thinning garlic mustard to 50% of its original cover) and no removal (Stinson et al. 2007). After 1 year, garlic mustard removal had no effect on native plant species richness, while native plant diversity increased in the full-removal treatment compared with the partial and no removal treatments (Stinson et al. 2007). Observation plots were also established to determine whether removal plots enabled full restoration of the native community. There was no difference in native species diversity between full-removal plots and uninvaded observation plots after 1 year (Stinson et al. 2007), but because all plots in the removal experiment were fenced while observation plots were not, the effects of weed absence and fencing were confounded.

In a study in Ohio, USA, spraying garlic mustard with glyphosate decreased wintergreen herb density after 2 years, but the most common wintergreen species were weeds (Carlson & Gorchov 2004). Garlic mustard removal also increased the cover of spring perennials in the old-growth stand (but not the second-growth stand), and had no effect on species richness, diversity or total plot cover after 2 years (Carlson & Gorchov 2004). After 4.5 years (with annual spraying), there was still no effect on species richness, diversity or composition, but the non-native wintergreen *Stellaria media* had lower cover in sprayed plots in the old-growth stand (Hochstedler et al. 2007). In sprayed plots, spring perennials tended to have higher cover in old-growth stands and annuals tended to have lower cover in second-growth stands, compared with untreated plots, after 4.5 years (Hochstedler et al. 2007). Adult garlic mustard density was lower in sprayed plots than untreated plots after 4.5 years (Hochstedler et al. 2007). In contrast, spraying had no effect on garlic mustard rosette abundance, which may have inhibited vegetation recovery in treated plots (Hochstedler et al. 2007). Continuous removal of garlic mustard in Maryland, USA, had no effect on understorey species richness or diversity after 3 years, but annuals tended to be more dominant in treated plots, and tree seedlings and vines were more abundant (McCarthy 1996).

Native species did not increase in response to removal of dame's rocket (*Hesperis matronalis*) in a north American forest, but there was an increase in the cover of exotic woody species (Pavlovic et al. 2009). Similarly, native vegetation did not recover 3.5 years after removal of invasive fennel (*Foeniculum vulgare*) on Santa Cruz Island, California, despite a significant reduction in fennel cover. Non-native grasses invaded plots where fennel was removed and probably inhibited native regeneration, along with the lack of a native seed bank (Erskine Ogden & Rejmánek 2005).

Rice et al. (1997) investigated the effects of controlling *Centaurea maculosa* using three herbicides at three sites with low levels of infestation in the USA, over a period of 7 years. Both once-sprayed plots and plots resprayed after 3 years had higher graminoid percentage cover and biomass than untreated plots, although plots that were sprayed only once had begun to return to untreated levels (Rice et al. 1997). Some of the herbicide treatments applied in early summer (before forbs had entered summer drought-induced dormancy) reduced plant species richness and/or diversity compared with untreated plots, while other early treatments and late-summer herbicide application had no effect (Rice et al. 1997).

Bracken fern (*Pteridium esculentum*) was controlled in heathlands using various combinations of cutting and herbicide application (Marrs & Lowday 1992). Cutting appeared to be the most effective, resulting in increased abundance of some native species after 10 years (Marrs & Lowday 1992). However, heathland restoration was not particularly successful, with seedlings of non-heathland tree species and weeds also establishing (Marrs & Lowday 1992).

Woody shrub and tree removal

Love & Anderson (2009) found significant differences in the effectiveness of four methods used to control Morrow's honeysuckle (*Lonicera morrowii*) in Pennsylvania, USA. Mechanical removal was the most effective at reducing Morrow's honeysuckle, while cutting shrubs and stump application of herbicide were least effective. For herbaceous plants, mechanical removal and cutting significantly increased total, native and exotic vegetation cover, and total, native and exotic species richness, while foliar herbicide application had no effect compared with untreated plots. Stump application of herbicide increased total herbaceous vegetation cover and exotic cover compared with untreated plots, but species richness did not differ. Native shrub cover decreased with both herbicide treatments but was not significantly affected by mechanical removal or cutting treatments. Preliminary results of a study examining best-practice chemical control of English broom (*Cytisus scoparius*) showed that species richness declined following chemical control, at least in the short term (Allan et al. 2006).

In another study of Amur honeysuckle (*Lonicera maackii*), this time in Ohio, USA, removal of the weed increased the survival of planted native seedlings compared with untreated plots (Hartman & McCarthy 2004). Similarly, in a study on the effects of controlling the invasive tree *Morella faya* on the establishment of native plants in Hawai'i, planted native seedlings only survived in plots where the weed had been controlled (Loh & Daehler 2007). In eastern North America, removal of invasive Norway maple (*Acer platanoides*) trees and saplings caused an increase in survivorship and density of native sugar maple (*Acer saccharum*) seedlings (Webb et al. 2001). Mason & French (2007) evaluated two different bitou bush (*Chrysanthemoides monilifera* ssp. *rotundata*) management regimes for their effects on native vegetation recovery in Australia: 'intensive', manually based weed control, and 'extensive' control (mainly herbicide spraying from aircraft). Results indicated that while biodiversity recovery was better at sites under 'intensive' management, such sites were also richer in other exotic species than 'extensively' managed sites (Mason & French 2007).

Multiple species removal

Removal of all exotic annuals from plots in Western Australian shrubland resulted in a threefold increase in the total percentage cover of native plants compared with untreated plots and increased seed production per flowering head in two native species (Hester & Hobbs 1992). One native species grew less well in plots where non-natives were removed but there were no other significant effects of non-native removal (Hester & Hobbs 1992). In a Hawaiian study, Wester (1994) reported that, despite an initial positive effect, removing multiple weed species had no long-term effect on the abundance of the threatened plant *Marsilea villosa*. In the Galapagos Islands, controlling multiple weed species increased native species abundance, while there was little recovery of native vegetation in untreated areas (Yanez et al. 2004).

Summary

In summary, the studies reviewed here are highly variable in terms of the species and systems studied, experimental designs employed, and what aspect of biodiversity was measured. Accordingly, it is difficult to generalise about the circumstances that lead to conservation gain following weed control. Failure to achieve conservation gain was common in the weed removal experiments included in this review—35% (18/51) of the experiments included at least one weed removal treatment that failed to achieve an improvement in any biodiversity variable/s measured (see Appendix 1). A similar number of studies (20/51) reported positive, negative and neutral biodiversity responses, depending on the treatment and/or variables measured. Only 9 out of the 51 studies reported a positive biodiversity response for all removal treatments. Although the reasons for failure were not always apparent, regeneration of the target weed and invasion by other weeds were common factors limiting success (see Table 1).

2.3 Weed addition experiments

The weed-addition approach involves introducing weeds into a previously uninvaded site and measuring the impacts. An advantage of weed additions is that the density of the weed can be manipulated easily, enabling impact thresholds to be determined (Adair & Groves 1998). Further, because pre-invasion states can be measured accurately, addition experiments can provide clear evidence for the impacts of weeds on biodiversity (Adair & Groves 1998). Weed addition experiments have been conducted under artificial conditions (for example, in pots in a glasshouse (not reported here)), but the approach has been used rarely in the field, where the most ecologically meaningful results could be obtained (Adair & Groves 1998). This is perhaps unsurprising given the potential problems likely to be associated with introducing, and containing, weeds in uninvaded areas. Certainly, this approach would not be acceptable on conservation land.

Table 1. Reasons why weed removal failed to achieve conservation gain and the type of supporting evidence (q = quantitative, s = speculative, – = no evidence). Studies where at least one removal treatment failed to have any positive effect on native biodiversity were included.

REASONS FOR FAILURE	STUDY	EVIDENCE
Invasion/recovery of same or other weed/s	Champion 2006a	q
	Cox & Allen 2008	q
	Erskine Ogden & Rejmánek 2005	s
	Hochstedler et al. 2007	q
	McAlpine et al. 2009	q
	Paul & Ledgard 2009	q
	Pavlovic et al. 2009	s
	Standish 2002	q, s
	Webb et al. 2001	s
	Williams et al. 1998	s
Control method failed to reduce weed abundance	Love & Anderson 2009	s
	Rinella et al. 2009	q
	Simmons et al. 2007	q
Native seed limitation	Erskine Ogden & Rejmánek 2005	s
	Loh & Daehler 2007	q
	Williams et al. 1998	s
Weed had no significant impact	Hejda & Pyšek 2006	q
	Turner & Virtue 2009	s
	Wester 1994	q
Not reported	Allan et al. 2006	–
	Bourdôt & Saville 2008	–
	Husted-Andersen 2002	–

Studies of planted species that later become weedy could be considered analogous to the weed addition approach. In a study of the impacts of weed invasion, Paul & Ledgard (2008b) documented vegetation changes over 30 years following wilding contorta pine (*Pinus contorta)* invasion in Canterbury, New Zealand. In the early stages of invasion, 26 native species were present, but by 2007 there were no native plants left (Paul & Ledgard 2008b). Several studies have also documented the native vegetation changes that occur when commercial crops of pines are planted. In New Zealand, Corsican pine (*P. nigra*) planting near Tekapo caused a decline in native species richness and cover compared with adjacent unplanted areas (Paul & Ledgard 2008b). In South Africa, 39 years after *Pinus radiata* had been planted, fynbos vegetation cover (excluding *P. radiata*) had declined from 75% to 20% and species richness had decreased by 58% (Richardson & van Wilgen 1986). Removal of *P. radiata* was not expected to fully restore native vegetation, as many of the eliminated species were poor dispersers that did not form a soil seed bank (Richardson & van Wilgen 1986). In a subsequent study in South Africa where *P. radiata* was felled and burned, species richness, native plant cover and density were all lower than in uninvaded sites after 15–16 months (Holmes et al. 2000).

3. How much control is required for conservation gain?

We were also interested in locating studies that investigated whether varying the level of control (e.g. all plants removed versus half of plants removed) resulted in different conservation gains. Identifying the minimum effort required to achieve conservation gain could help weed managers to maximise the cost effectiveness of their weed control programmes. The degree of weed control achieved can vary depending on the control methods used (e.g. Williams et al. 1998; Cabin et al. 2002; Love & Anderson 2009), the timing of control (Allan et al. 2006), the frequency of follow-up control, and environmental conditions (Carlson & Gorchov 2004).

Stinson et al. (2007) reported that native tree seedling abundance increased in response to partial removal (weeding to c. 50% of original cover) of garlic mustard, but did not increase in response to full removal. Conversely, native species richness did not increase in response to either treatment, and the authors concluded that the effectiveness of full or partial weed removal depends on what outcome is desired (Stinson et al. 2007). Loh & Daehler (2007) tested the effect of variation in kill rates of the invasive tree *Morella faya* on the establishment of native plants in Hawai‘i. *Morella faya* was controlled by logging all trees, leaving trees standing but girdled (ring-barked) and incremental girdling over 20 months. Untreated control plots were also established. Seedlings of three native pioneer species and three native forest species were planted in plots 28 months after stands were treated. No planted native pioneer or forest seedling species survived in the untreated plots. Survivorship of planted pioneer native seedlings was highest in girdle and log treatments (52% and 48% respectively) and lowest in incremental girdle plots (22%). Survival of native forest species did not differ among stand treatments (Loh & Daehler 2007).

Scientists at Landcare Research and DOC are currently investigating whether weeds facilitate native tree and shrub establishment in New Zealand dryland environments (Walker et al. 2009). Scotch broom (*Cytisus scoparius*), gorse and blackberry (*Rubus fruiticosus*) shrublands were treated by spraying, roller crushing, mulching and root-raking, then native seeds and seedlings were added to plots. Results to date suggest that both seed germination and seedling survival of native species is higher in plots where some weed cover remains (untreated plots and roller-crushed treatments) than they are in plots where all weed cover is removed (mulch and root-rake treatments) (Larry Burrows, Landcare Research, Lincoln, pers. comm. 2009).

In eastern North America, removal of invasive Norway maple (*Acer platanoides*) trees and saplings caused an increase in survivorship and density of native sugar maple (*Acer saccharum*) seedlings and decreased Norway maple seedling recruitment (Webb et al. 2001). In contrast, removal of only Norway maple seedlings caused a significant pulse in Norway maple seedling recruitment but not in recruitment of sugar maple seedlings (Webb et al. 2001). In a Californian study, hand weeding English ivy resulted in a significant increase in native seedling density, diversity and species richness compared with spraying with herbicide, while no seedlings were found in any untreated plots (Biggerstaff & Beck 2007). Treatment effects did not appear to be due to differences in the level of control achieved, as both control methods apparently removed all ivy for the duration of the experiment (7 months), although this was not specifically stated. Differences between treatments were thought to be due to the presence of dead ivy inhibiting regeneration in sprayed plots (Biggerstaff & Beck 2007).

MacDougall & Turkington (2005) found no significant differences in community structure or functional group responses between weeding and mowing for two invasive grasses in Canada. Allan et al. (2006) reported early results from an adaptive experimental management programme for English broom in Victoria, Australia. The programme was designed to evaluate the effectiveness, efficiency and environmental outcomes of two chemical sprays applied in spring and autumn. Initial results showed a decrease in broom cover and plant species richness in treated plots compared with untreated controls after 2 years (Allan et al. 2006). Broom cover did not appear to differ between the two sprays, but autumn spraying was more effective than spring spraying (Allan et al. 2006).

The rate at which weeds are killed can also influence the regeneration of both native plants and weeds. The sudden death of established woody weed species can promote rapid establishment of other invaders in forest ecosystems (Webb et al. 2001; Loh & Daehler 2007), while slower mortality rates may provide suitable conditions for natives to establish (Loh & Daehler 2007). Weed species often germinate more rapidly than natives (Pérez-Fernández et al. 2000; Loh & Daehler 2007), so the high light conditions created by clear-felling can promote the establishment of weeds, which then impede the subsequent establishment of natives (Loh & Daehler 2007). However, early germination is not always advantageous (Kelly 1989), and native species may have an edge in dense shade under an intact canopy (Loh & Daehler 2007) and may regenerate as more short-lived weeds die.

In summary, there have been very few studies that specifically test the effect of varying levels of weed control on native biodiversity. Many studies compare the effects of different weed removal methods (e.g. hand weeding versus herbicide, or the use of different herbicides) on native biodiversity, rather than the effects of specifically varying weed infestation levels (e.g. Williams et al. 1998; Hartman & McCarthy 2004; MacDougall & Turkington 2005; Allan et al. 2006; Flory 2010). The one study that specifically compared two different levels of weed removal using the same treatment (hand-weeding) concluded that the effectiveness of full or partial weed removal depended on what aspect of native biodiversity was being measured (Stinson et al. 2007). Some studies applied the same treatment during different seasons, which happened to result in different levels of weed control. However, it can be difficult to infer an effect of weed abundance on native biodiversity due to the confounding effects of treatment method. For example, although weed removal may be beneficial for native vegetation, herbicides can sometimes kill non-target species, thus having a negative effect; while a similar level of weed control using manual removal may be beneficial.

4. Modelling weed impacts and the concept of weed control thresholds

'A key consideration in managing weeds in natural ecosystems lies in deciding at what point intervention should occur and to what extent' (Panetta & James 1999: 68). Henry (1994, cited in Panetta & James 1999) suggested that environmental weeds could be managed more effectively by attempting to keep weed populations below a threshold that would cause 'native plant loss or other ecosystem degradation'. Ecological thresholds can be broadly defined as points or zones at which relatively rapid change occurs from one ecological condition to another (Huggett 2005). Other authors have also recognised the potential for using weed impact thresholds to prioritise or optimise weed control efforts (Byers et al. 2002; Yokomizo et al. 2009). Adair & Groves (1998) proposed that weed threshold levels for declines in biodiversity could be used to set the maximum tolerable level of weed infestations. They argued that reducing weed levels to well below such thresholds will probably be a poor use of resources, except where the aim is eradication. Although weed control thresholds are an intuitively appealing concept, their utility in managing environmental weeds is uncertain. Despite widespread acceptance of the weed threshold concept in agricultural ecosystems (Adair & Groves 1998) and more than 30 years of research on thresholds and crop damage, almost no farmers use weed control thresholds to manage weeds in crops (Panetta & James 1999). There appears to have been little research on impact thresholds for environmental weeds. However, scientists at Landcare Research are currently investigating both above- and below-ground impacts of various weed species in New Zealand, and are examining whether these impacts vary with weed abundance (Duane Peltzer, Landcare Research, Lincoln, pers. comm.).

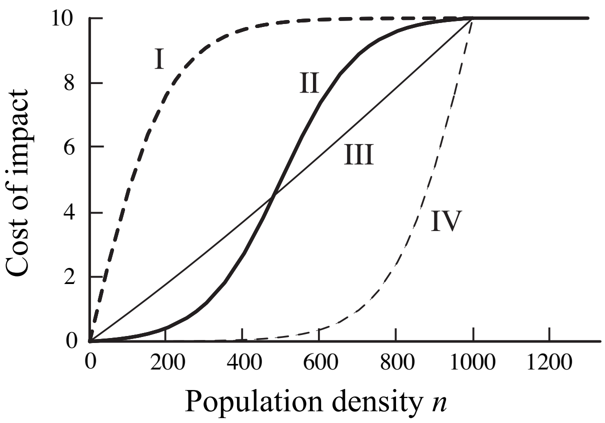

Figure 1. Four potential relationships between the cost of impact and weed population density: I, low-threshold curve; II, S-shaped curve; III, linear curve; IV, high-threshold curve (reproduced with permission from Yokomizo et al. (2009)).

Defining the relationship between weed abundance and ecosystem degradation is critical to determining the optimal management effort (Yokomizo et al. 2009) and whether weed impact thresholds are a useful management tool for a given weed species. Several models of the relationship between weed infestation levels and biodiversity impact have been developed. Yokomizo et al. (2009) identified four potential relationships between weed density and the degree of impact, termed density-impact curves (see Fig. 1). In the low-threshold curve (curve I) the impact remains high until the weed population density becomes very low. In contrast, in the high-threshold curve (curve IV) the impact remains low and then increases dramatically only when the population density is very high. Curve II is an S-shaped curve with the impact increasing rapidly at an intermediate population density, while in curve III (linear relationship) the impact is directly proportional to population density (Yokomizo et al. 2009). Curves I, II and IV are all forms of a threshold relationship between weed density and impact. The form of the density-impact curve may differ within a weed species, between different habitats, or with different measures of impact (Yokomizo et al. 2009).

Other models that can explain the relationship between weed infestation and impact include a hyperbolic relationship, which is often found between percentage crop yield losses and weed density. The hyperbolic model has a sound biological basis given that the size of individual weeds (and thus their competitive abilities) are at a maximum at low densities (Panetta & James 1999). Weeds can also have a beneficial effect (described by a positive linear or curved model) or no effect (indicated by a horizontal line) (Adair & Groves 1998). However, neither of these models is of interest in determining weed impact thresholds and they are not discussed further.

A negative linear or uniformly curved relationship predicts that any level of weed control will result in an improvement for biodiversity (Adair & Groves 1998). Conversely, this type of relationship also indicates that some level of damage will occur at any weed density above zero. This type of relationship makes it difficult to define a useful threshold objectively, and weed managers will need to decide what level of impact they are willing to tolerate (or what resources are available) prior to implementing a control programme. For example, the management goal might be to restore 90% of the original plant species present in a weed-infested ecosystem. To achieve this outcome, we need to calculate the slope of the linear relationship between weed infestation and native plant species richness. The regression slope quantifies the strength of the effect, with a steeper slope indicating greater intensity (Aarssen & Epp 1990), and is critical to determining the degree of control required (Panetta & James 1999). Alternatively, we may be interested in managing a weed to protect a threatened plant species. In this case, we need to determine the relationship between weed abundance and persistence and/or abundance of the threatened species. Yokomizo et al. (2009) discuss optimal weed management efforts in relation to the density-impact curves.

Many studies report a negative linear relationship between weed abundance and native plant species richness and/or seedling density (Braithwaite et al. 1989; Woods 1993; Hutchinson & Vankat 1997; Abensperg-Traun et al. 1998; Alvarez & Cushman 2002; Jackson 2005; Oswalt et al. 2007; Stephens et al. 2008), including two from New Zealand (Kissling et al. 2005; Barton et al. 2007). Kissling et al. (2005) found a negative linear relationship between cover of the invasive tussock grass *Nardus stricta* and native species richness and cover at Lake Pukaki, Canterbury. There was also a negative relationship between native species richness and the percentage cover of two invasive species—mist flower and selaginella (*Selaginella kraussiana*)—at several North Island sites (Barton et al. 2007).

There may also be a negative relationship between weed abundance and the abundance of native fauna. For example, in an Australian study, Braithwaite et al. (1989) found a decrease in the abundance of a number of native mammal, bird and lizard species with increasing density of the invasive species *Mimosa pigra* (although not all animal species were negatively affected by this invasive weed). In contrast, McCarthy (1996) found no evidence for a linear relationship between biomass of the invasive garlic mustard and species richness in an eastern USA oak forest.

Some relationships may be better described by non-linear models. For example, McConkey & Drake (2006) found that although a linear regression model showed a significant fit to their data for seed dispersal distance and flying fox abundance (r^2 = 0.72), the relationship was much stronger when a non-linear piecewise regression model was fitted (r^2 = 0.92). Although not an example of the relationship between weed abundance and conservation gain, this study illustrates the importance of using appropriate statistical analyses. Braithwaite et al. (1989) reported a negative linear relationship between *Mimosa pigra* density and both herbaceous species richness and native tree seedling density, but r^2 values were low (0.078 and 0.325, respectively). The relationship may have been better explained by an alternative model.

Threshold models are characterised by a negative linear or curved relationship with a well-defined change of slope (Adair & Groves 1998). The change of slope represents the threshold value of weed infestation impacts on biodiversity and is the minimum point at which control measures should be targeted (Adair & Groves 1998). Determining weed impact thresholds can be problematic and they are sometimes defined rather arbitrarily; for example, when no clear threshold can be demonstrated or the relationship is poorly defined (Panetta & James 1999). In agriculture, weed thresholds tend to be used in an economic sense to calculate whether the benefits from control outweigh the costs of the operation and any toxic effects on the crop (Cousens & Mortimer 1995). The invasive agricultural weed Paterson's curse (*Echium plantagineum*) has a high-threshold density-impact curve (see Fig. 1, curve IV). Paterson's curse is toxic to livestock, which avoid it at low density, thus its impact is minimal except at high population densities (Seaman et al. 1989, cited in Yokomizo et al. 2009).

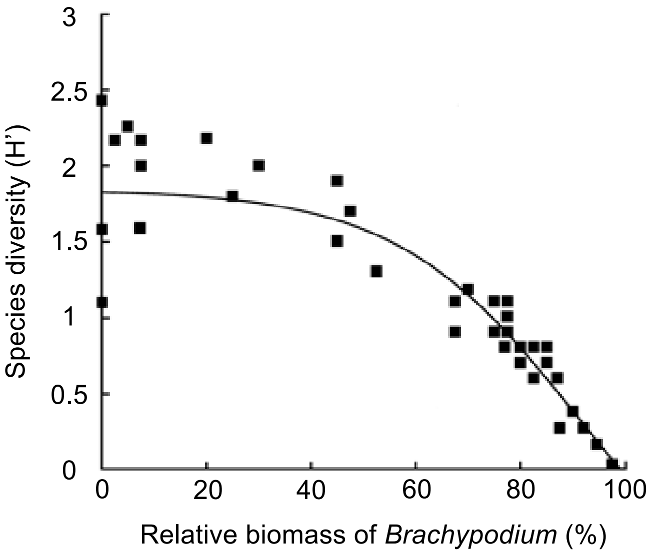

Figure 2. Relationship between species diversity and the relative phytomass of *Brachypodium pinnatum* in chalk grasslands (reproduced with permission from Panetta & James (1999), redrawn from Bobbink & Willems (1987)).

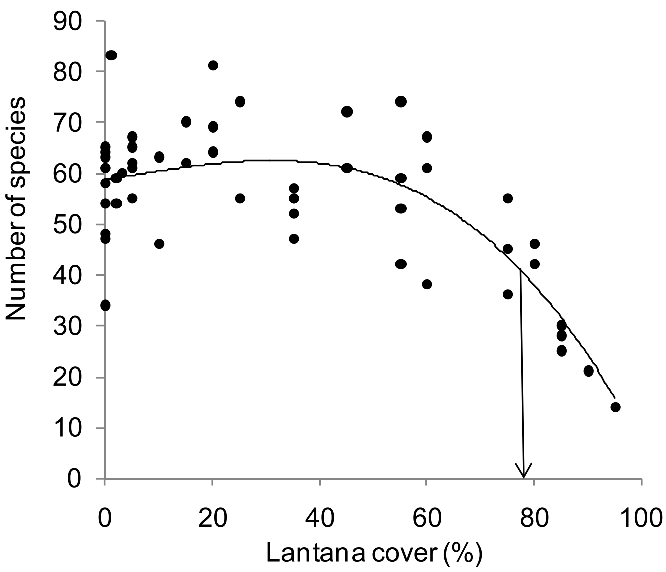

Figure 3. Relationship between total native species richness and percentage cover of *Lantana camara* (reproduced with permission from Gooden et al. (2009)).

Bobbink & Willems (1987) determined the relationship between biomass of the invasive weed *Brachypodium pinnatum* and native species diversity in chalk grasslands of western Europe. They contended that *Brachypodium* started to influence biodiversity once it reached more than 50% of the above-ground biomass. This was based on a strong negative linear relationship (r^2 = 0.815, P = 0.001) between *Brachypodium* biomass and diversity where *Brachypodium* biomass was greater than 50%. When Panetta & James (1999) reanalysed the entire dataset, they fitted a cubic relationship and found no threshold level of impact (see Fig. 2). They therefore argued that an impact threshold of 50% was rather subjective, and noted that the presence of *Brachypodium* was associated with a decline in diversity at even low biomass levels. However, Panetta & James (1999) did not report r^2 or P-values to indicate how well the cubic model fitted the data and did not fit any alternative models. In addition, the rate of decline in species diversity was much greater at *Brachypodium* biomass levels greater than 50% (indicated by the increase in the steepness of the slope in Fig. 2). The data could well fit a threshold model, characterised by a well-defined change of slope in the relationship between weed biomass and species diversity (Adair & Groves 1998). Appropriate statistical methods for identifying ecological thresholds are discussed in Toms & Lesperance (2003).

Gooden et al. (2009) got a similar-shaped curve when analysing the relationship between percentage cover of the invasive species *Lantana camara* (lantana) and total native species richness in wet sclerophyll forest in southeastern Australia (see Fig. 3). The significant negative relationship between lantana cover and total native species richness was non-linear, with the rate of species loss increasing with lantana invasion. The authors identified a potential lantana impact threshold of around 80% lantana cover (Gooden et al. 2009).

Alvarez & Cushman (2002) reported a significant decrease in native plant species richness with increasing Cape ivy cover in invaded plots (see Fig. 4; r^2 = 0.505–0.616). In this case, a threshold relationship seems likely as mean species richness in uninvaded plots was considerably lower than would be predicted by the regression analysis for invaded plots (Alvarez & Cushman 2002). However, the authors were unable to determine the specific shape of the relationship and whether an impact threshold exists because there were no data on species richness for Cape ivy cover values between 0% and 47% (Alvarez & Cushman 2002). Reinhardt Adams & Galatowitsch (2006) claimed a threshold effect between reed canary grass biomass and the biomass of other vegetation, but did not report any statistical evidence to support this.

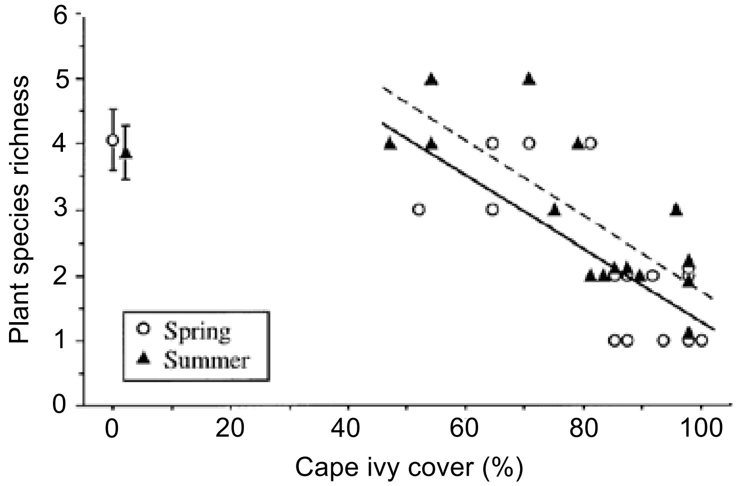

Figure 4. Relationship between native plant species richness per plot and percentage cover of Cape ivy (*Delairea odorata*) during spring (circles and solid line) and summer (triangles and dashed lined). Also presented (at left of data field) are data on plant species richness (mean ± 1 SE) for uninvaded plots at these two times in the growing season (reproduced with permission from Alvarez & Cushman (2002)).

Research on *Tradescantia fluminensis* provides arguably the best New Zealand example to date demonstrating a weed control threshold required for restoration of native vegetation (Standish et al. 2001). In a field study at three forest remnants in Manawatu, *Tradescantia* biomass increased with increasing light levels, and native forest seedling species richness and abundance decreased exponentially as *Tradescantia* biomass increased (Standish et al. 2001). At one site, *Dysoxylum spectabile* seedlings survived for 20 months only when *Tradescantia* cover was below a threshold of c. 200 g/m^2 (equivalent to 70–90% cover) (see Fig. 5; Standish et al. 2001). Some native seedlings were found at *Tradescantia* biomass levels up to c. 400 g/m^2 and occasionally up to c. 550 g/m^2 (Standish et al. 2001). Whether seedlings of species other than *D. spectabile* can survive where *Tradescantia* biomass is greater than 200 g/m^2 is unknown.

In summary, weed impact thresholds are apparent from some studies but not others, and are, therefore, likely to be highly variable across species and sites. Thus, it is difficult to summarise how they might be used to determine the stage at which weed populations must be controlled in order to avoid significant negative impacts.

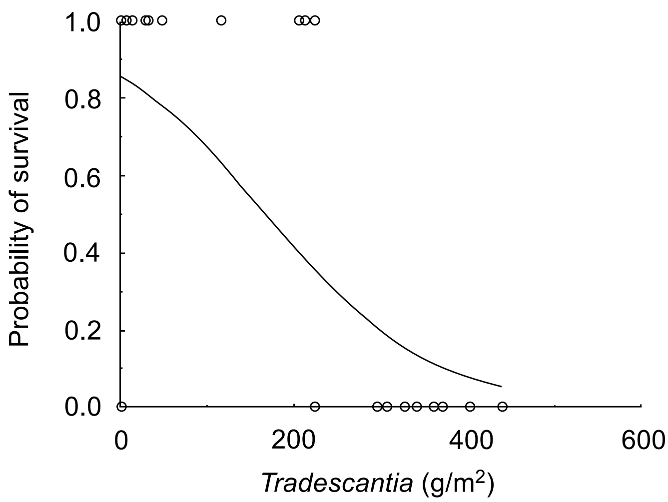

Figure 5. Logistic regression model of survival probability for *Dysoxylum spectabile* seedlings (reproduced with permission from Standish et al. (2001), Wiley-Blackwell).

5. Effects of post-control management on native biodiversity

Weed control alone is not always sufficient to restore native plant communities and additional management intervention may be necessary (D'Antonio & Meyerson 2002; Turner et al. 2008). Weed control often promotes invasion by the same or other weed species through soil disturbance, increased light and nutrients, and trampling (Alvarez & Cushman 2002; D'Antonio & Meyerson 2002; Erskine Ogden & Rejmánek 2005; Hulme & Bremner 2006; Loh & Daehler 2007; Mason & French 2007; Love & Anderson 2009; McAlpine et al. 2009; Pavlovic et al. 2009). Sites with a long history of human impact or weed invasion (or in close proximity to such areas) will probably have seed banks dominated by introduced species (Hester & Hobbs 1992; Luken 1996). Weed management should attempt to alter the successional trajectory of invaded sites by encouraging desirable species as well as discouraging weeds (Luken 1996). To this end, scientists at DOC are currently investigating how to control particular weed species and life forms in order to facilitate native plant succession.

Seed supply often limits native plant recruitment after weed control (Marrs & Lowday 1992; Cabin et al. 2002; Suding et al. 2004; Biggerstaff & Beck 2007; Loh & Daehler 2007; Cox & Allen 2008). Loh & Daehler (2007) added seeds of three native pioneer species, three native forest species and three weed species to plots 14 months after controlling the invasive species *Myrica faya*. Recruitment of native species after 1 year was limited by seed availability, as no seedlings emerged from plots unless seeds had been added. Cox & Allan (2008) also reported an increase in native forb percentage cover when seeds were added compared with unseeded plots, after exotic grass control. In contrast, Flory (2010) reported that native seed addition after *Microstegium vimineum* control decreased resident plant productivity (biomass) compared with plots without seed addition.

Grazing by livestock and feral mammals can also limit native plant regeneration after weed control (Abensperg-Traun et al. 1998) and fencing to exclude grazers may be necessary (Cabin et al. 2000). Monitoring and maintenance after weed removal and restoration is essential to the success of any control programme. Weeds with a soil seed bank or extensive, persistent rhizomes may require repeated follow-up treatment, sometimes for many years (D'Antonio & Meyerson 2002).

6. Summary

Despite extensive environmental weed control efforts both in New Zealand and other countries, few studies have quantified whether weed control actually achieves conservation gain. This does not necessarily mean that weed control is a waste of time and does not deliver conservation gain. It may merely reflect the fact that such studies are rarely undertaken. The lion's share of weed management money tends to be used for killing the weed, with little left for expensive post-control monitoring of native biodiversity. Additionally, individual sites require individual management regimes, so it is difficult to identify commonalities on which to base generalisations. Indeed, the studies we report here show that conservation gains following weed control are highly variable. Many weed control programmes fail to achieve any conservation gain, and many have both positive and negative effects on biodiversity, depending on the treatment method used and/or response being measured. The explanations for why weed control programmes fail to achieve conservation gain are also highly variable, often purely speculative or, in many cases, unknown. Accordingly, it is difficult to identify common factors contributing to the success or failure of weed control programmes. It is also difficult to determine what level of control is required for conservation gain, given the scarcity of studies that have investigated this question to date. The use of weed impact thresholds to determine when weeds must be controlled in order to avoid negative impacts is an intuitively useful concept, but again, data are insufficient for it to be of much practical use.

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Appendix 1. Summary of weed removal experiments. Response of the biodiversity variable measured is categorised as positive (+) if it improved following weed removal (e.g. plant species richness increased), negative (–) if it decreased, or neutral (0) if it did not change. Where more than one removal treatment was used, results are presented in the same order as numbered removal method

AUTHOR AND COUNTRY OF STUDY	STUDY AIM	WEED SPECIES STUDIED	BIODIVERSITY VARIABLE MEASURED	LENGTH OF STUDY	NO. SITES	NO. REPLICATES	PLOT SIZE	REMOVAL TREATMENTS	LEVEL OF CONTROL ACHIEVED	FOLLOW-UP CONTROL
Allan et al. 2006; Australia	Effect of weed removal on weed cover and plant species richness	English broom (*Cytisus scoparius*)	Plant species richness (–, –, –)	2 years	3	Unclear	49 cm diameter	1. Selective herbicide 2. Non-selective herbicide 3. Spring or autumn application 4. Untreated	c. 20–90%	Annually
Alvarez & Cushman 2002; USA	Impact of weed on native plant community in three habitat types	Cape ivy (*Delairea odorata*)	Native seedling abundance (0) Native species richness (+) Total diversity (+) Forb species richness (+) Fern species richness (0) Grass species richness (0) Shrub species richness (–)	2 years	3	4 blocks each at two sites and 2 blocks at one site	1.5 m^2	1. Hand weeded 2. Untreated	50% cover after 2 years (79–82% cover in untreated plots)	1 month after initial control and 1 year after first follow-up
Barton et al. 2007; New Zealand	Assess agent establishment and dispersal, measure attack levels, and monitor changes in native and exotic vegetation	Mist flower (*Ageratina riparia*)	Native species richness (+, +) Native species % cover (+, +)	5 years	1	10	4 m^2	Biocontrol agents: 1. White smut fungus (*Entyloma ageratinae*) 2. Gall fly (*Procecidochares alani*)	c. 75% cover in 1999/2000 v. c. 2% cover in 2003/2004	None
Biggerstaff & Beck 2007; USA	Effect of ivy removal methods on soil chemistry and native regeneration	English ivy (*Hedera helix*)	Native seedling: density (+, +) species richness (+, +) species diversity (+, +)	7 months	1	5	15.5 m x 5 m	1. Leaves cut and sprayed with glyphosate 2. Hand-weeded 3. Untreated	Not reported	Sprayed plots re-sprayed after 2 weeks
Bourdôt & Saville 2008; New Zealand	Effectiveness of herbicide and tolerance of *Lepidium kirkii* to herbicide	*Plantago coronopus*	Native *L. kirkii* density (0, 0)	3 years	3	8	0.5 m^2	1. 2,4-DB 2. Versatill™ 3. untreated	1. 0.6 plants/plot 2. 0.7 plants/plot 3. 13.7 plants/plot	Annually
Cabin et al. 2002; Hawai‘i	Investigate factors affecting native establishment and invasive grass suppression	Fountain grass (*Pennisetum setaceum*)	Native % cover (+, +, +, +)	18 months	1	4	6 m^2	1. Bulldoze 2. Herbicide 3. Mulch 4. Trim	Approximate cover 1. 29% 2. 35% 3. 34% 4. 80%	None
Carlson & Gorchov 2004 and Hochstedler et al. 2007; USA	Impact of garlic mustard on native community and effectiveness of herbicide control; effect of removal on native community	Garlic mustard (*Alliaria petiolata*)	Species richness (0) Species diversity (0) Species composition (0) % cover wintergreen herbs (–) % cover other growth forms (0)	4.5 years	2	25 per treatment, per site	1 m^2	1. Sprayed with glyphosate 2. Untreated	11–29 plants/ m^2 in treated plots v. 32–36 in untreated plots	Resprayed annually

Continued on next page

Appendix 1 continued

AUTHOR AND COUNTRY OF STUDY	STUDY AIM	WEED SPECIES STUDIED	BIODIVERSITY VARIABLE MEASURED	LENGTH OF STUDY	NO. SITES	NO. REPLICATES	PLOT SIZE	REMOVAL TREATMENTS	LEVEL OF CONTROL ACHIEVED	FOLLOW-UP CONTROL
Champion 2006b; New Zealand	Effect of grass removal on native plant community	Reed canary grass (*Phalaris arundinacea*)	Native sedge cover (+) Native woody cover (+) Native aquatic plant cover (+)	1 year	1	4	10 m^2	Sprayed with Gallant™ herbicide	Mean 6.8% cover (range 0–22.7%) cf. 84.6% before treatment	None
Cox & Allen 2008; USA	Effects of grass control methods on exotic grasses, exotic forbs and native forbs	Exotic grasses (*Avena fatua*, *Bromus diandrus*, *B. hordeaceus*, *B. madritensis* ssp. *rubens*)	Native forb cover (0, 0, 0)	6 years	1	5	10 m^2	1. Herbicide 2. Thatch removal + herbicide 3. Mowing 4. Untreated	Approx. cover 1. 15% 2. 20% 3. 26% 4. 34%	Herbicide reapplied after 1 year; annual mowing for 3 years
D'Antonio et al. 1998; Hawai'i	Effect of bunchgrass removal on native species	Bunchgrass (*Schizachyrium condensatum*)	Native shrub growth (+) Native woody seedling density (+) Shrub architecture (+) Soil and native plant leaf nitrogen (+)	3 years	1	3	20 m^2	1. Hand weeded by chopping plants off below root crown 2. Untreated	Not reported	Every 6 months
Daehler & Goergen 2005; Hawai'i	Feasibility of restoring indigenous grassland where buffel grass is dominant	Buffel grass (*Cenchrus ciliaris*)	Native *Heteropogon* grass % cover (0, +, +)	4.5 years	1	1998: 9 burned, 8 hand-pull + burn, 8 burn + herbicide, 3 untreated. 2000: 15 burn + herbicide, 3 untreated	1 m^2	1. Burn 2. Hand pull + burn 3. Burn + herbicide 4. Untreated	1. <1 seedling/m^2 2. <1 seedling/m^2 3. <1 seedling/m^2 4. 10 seedlings/m^2	Repeated burning every 2 years
Erskine Ogden & Rejmánek 2005; Santa Cruz Island, USA	Effect of weed removal on native community	Fennel (*Foeniculum vulgare*)	Plant community structure (0) Native species cover (0) Native species richness (0) Native species diversity (0)	3.5 years	1	5 untreated grassland, 5 treated grassland, 5 untreated fennel, 10 treated fennel	30 m^2	1. Burned then aerially sprayed with triclopyr 2. Untreated	1. <3% cover 2. c. 28–51% cover	Aerially sprayed 6 months and 18 months after burning
Flint & Rehkemper 2002; Laysan Island*	Eradication of sandbur	Sandbur (*Cenchrus echinatus*)	Native grass *Eragrostis variabilis* % cover (+) Seabird nest density (+)	8 years	1	5 transects, 12 plots	150–300 m transects; 10 m^2 plots	Sprayed with glyphosate and hand-weeded	84.7 plants/hour detected after 3 years and 0.043 plants/hour after 8 years	Hand-weeded every 2 weeks, then every 6 weeks for 3 years after last plant found, then every 16 weeks if no new plants found; annual checks if no new plants in 5 years

* Weed control undertaken as part of conservation management programme and not as a specifically designed removal experiment.

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AUTHOR AND COUNTRY OF STUDY	STUDY AIM	WEED SPECIES STUDIED	BIODIVERSITY VARIABLE MEASURED	LENGTH OF STUDY	NO. SITES	NO. REPLICATES	PLOT SIZE	REMOVAL TREATMENTS	LEVEL OF CONTROL ACHIEVED	FOLLOW-UP CONTROL
Flory 2010 and Flory & Clay 2009; USA	Effect of removal methods on weed and native community	Japanese stilt grass (*Microstegium vimineum*)	Native vegetation biomass (+, +, 0) Native cover (+, +, 0) Native diversity (+, +, 0) Native grass richness (+, 0, –) Native forb richness (+, +, 0) Native woody richness (0, 0, 0) Native fern richness (0, 0, 0) Native grass biomass (0, –, 0) Native forb biomass (+, +, +) Native woody species biomass (0, 0, 0) Native fern biomass (0, 0, 0) Native grass % cover (+, 0, 0) Native forb % cover (+, +, +)	2 years	8	10 per treatment per site	2 m^2	1. Hand-weeded 2. Grass-specific post-emergent herbicide 3. Post-emergent herbicide + pre emergent herbicide 9 months later 4. Untreated	1. Biomass reduced by 87% 2. 99% reduction 3. 99% reduction	All treatments reapplied 1 year after their initial application
Hájková et al. 2009; Czech Republic	Assess most appropriate measure to restore *Molinia*-invaded fens	Tall moor grass (*Molinia arundinacea*)	Vascular plant species richness (+, +, 0) Bryophyte species richness (+, +, 0) Bryophyte cover (+, +, 0) Evenness (+, +, 0) Species composition (+, +, 0)	5 years	1	5	1 m^2	1. Annual mowing 2. Mowed twice a year 3. Annual litter removal 4. Untreated	Not reported	Annually or 6-monthly
Hartman & McCarthy 2004; USA	Effectiveness of control methods for survival of planted native seedlings	Amur honeysuckle (*Lonicera maackii*)	Planted native seedling survival (+, +) Native seedling growth (0, 0)	2.5 years	1	8	4.5 m x 5.5 m	1. Cut and painted 2. Injected and left standing 3. Untreated	94–95% mortality in both removal treatments	None
Hejda & Pyšek 2006; Czech Republic	Impact of weed on native community	Himalayan balsam (*Impatiens glandulifera*)	Species: richness (0) diversity (0) evenness (0) composition (0)	3 months	1	10	1 m^2	1. Removed 2. Untreated	Not reported	Removed for 3 months prior to monitoring
Hester & Hobbs 1992; Australia	Effects of fire on community dynamics and invasibility	All annual weed species present	Native annual % cover (+, –) Total native plant cover (+, +) Native species plant size (0, 0) Native seed production (+, +)	6 weeks	1	16	1 m^2	1. Weeded, presumably by hand in: a. shrubland and b. woodland 2. Untreated	Complete removal	Every 2 weeks
Hulme & Bremner 2006; United Kingdom	Impact of weed on native community	Himalayan balsam (*Impatiens glandulifera*)	Total species abundance (+) Total species richness (+) Total species diversity (+) Total seedling density (+) Total species evenness (–) Native species richness (+) Native species diversity (+) Native species evenness (0)	4 months	1	12	1 m^2	1. Cutting stems at ground level and removing 2. Untreated	100%	None

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AUTHOR AND COUNTRY OF STUDY	STUDY AIM	WEED SPECIES STUDIED	BIODIVERSITY VARIABLE MEASURED	LENGTH OF STUDY	NO. SITES	NO. REPLICATES	PLOT SIZE	REMOVAL TREATMENTS	LEVEL OF CONTROL ACHIEVED	FOLLOW-UP CONTROL
Husted-Andersen 2002; New Zealand	Non-target effects of control methods and effectiveness of aerial spraying on grey willow	Grey willow (*Salix cinerea*)	Species richness (0, 0) Species diversity (0, 0) Proportion native:exotic species (0, 0) Total plant abundance (–, 0) Native plant abundance (–, 0)	1 year	1	7?	2 m^2	1. Aerially sprayed with glyphosate 2. Cut and picloram hand-applied to stump	1. 69% killed 2. Not reported	None
Loh & Daehler 2007; Hawai'i	Effect of fire tree kill rate on native and invasive plant establishment	Fire tree (*Morella faya*)	Native seedling emergence: *Dodonea viscosa* (+, 0, 0) *Vaccinium reticulatum* (0, 0, 0) *Machaerina angustifolia* (+, +, +) *Ilex anomala* (0, +, +) *Vaccinium calycinum* (0, +, 0) *Isachne distichophylla* (+, +, +) Native seedling survival (+, +, +) Native seedling growth (+, +, +)	37 months	1	5 edge and 5 interior	30 m^2	1. Clear felled and stump painted 2. Ring-barked and herbicide applied to wound 3. Ring-barked and herbicide applied to trees in three lots at 0, 12 or 20 months 4. Untreated	Not reported Presumably 100% by end of study	none
Love & Anderson 2009; USA	Effects of removal methods on weed and native community	Morrow's honeysuckle (*Lonicera morrowii*)	Native herb % cover (+, 0, +, +) Native shrub density (0, –, –, 0) Native species richness (+, 0, 0, +)	15 months	1	10 per treatment and 5 control plots	5 m^2	1. Mechanical removal 2. Foliar glyphosate application 3. Stump glyphosate application 4. Cut at ground level 5. Untreated	1. 3% cover 2. 10% cover 3. 22% cover 4. 26% cover 5. 90% cover	None
MacDougall & Turkington 2005; Canada	Causes of weed dominance	*Poa pratensis* and *Dactylis glomerata* (*Cytisus scoparius* seedlings also removed after initial weeding)	Species richness (+, +) Species diversity (+, +) Species composition (+, +) Native forb % cover (+, +) Non-obligate native forb % cover (–, –)	4 years	2	10	1 m^2	1. Hand weeded 2. Mowed 3. Untreated	1. *Poa* c. 2–22% cover; *Dactylis* < 2% 2. *Poa* < 15%; *Dactylis* c. 2–12% 3. *Poa* c. 52%; *Dactylis* 17–28%	Annually
Maloney et al. 1999; New Zealand	Effect of willow removal on bird habitat use	Crack willow (*Salix fragilis*)	Native bird density and nest abundance (+)	3 years	3	1 removal site and 2 untreated sites (1 without willows and 1 with willows)	1.5–2.5 m x 1–1.5 km	1. Mechanically dug out and burned 2. Untreated	100% killed	None
Marrs & Lowday 1992; United Kingdom	Effectiveness of control and restoration treatments	Bracken (*Pteridium aquilinum*)	Cut twice yearly: *Agrostis capillaris* abundance (+, +, +, +, +) *Dicranium scoparium* (+, +, +, +, +) *Galium saxatile* (+, +, +, +, +) *Calluna vulgaris*: Cut once (+, +, +, +, +) All other treatments (+, +, +, +, +)	10 years	2	4	1 m^2	1. Cut yearly 2. Cut twice yearly 3. Asulam applied once 4. Asulam applied twice 5. Asulam applied once and yearly cutting thereafter 6. Untreated	Not reported	Varied among treatments

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AUTHOR AND COUNTRY OF STUDY	STUDY AIM	WEED SPECIES STUDIED	BIODIVERSITY VARIABLE MEASURED	LENGTH OF STUDY	NO. SITES	NO. REPLICATES	PLOT SIZE	REMOVAL TREATMENTS	LEVEL OF CONTROL ACHIEVED	FOLLOW-UP CONTROL
Mason & French 2007; Australia*	Effect of management regimes on plant community	Bitou bush (*Chrysan-themoides monilifera* ssp. *rotundata*)	Foredune: native species richness (0, 0) native vegetation composition (+, +) Hind-dune: native species richness (0, 0) native vegetation composition (0, 0)	Variable, usually <10 years	65 fore-dune; 34 hind dune	Foredune: 17 non-invaded, 19 intensively managed, 13 extensively managed, 16 invaded	20 m x 50 m; 20 m^2	1. Intensive: hand removal and herbicide applied directly 2. Extensive: herbicide sprayed aerially	3.3–16.6% cover (59.4–64.6% in untreated sites)	Variable, at least every 2 years
McAlpine et al. 2009; New Zealand	Effect of bone-seed control on bone-seed and native regeneration	Bone-seed (*Chrysan-themoides monilifera* ssp. *monilifera*)	Native seedling abundance (–)	4 months	4	40	1 m^2	Plants cut at ground level and removed	Not reported	None
McCarthy 1996; USA	Impact of garlic mustard on forest understorey community	Garlic mustard (*Alliaria petiolata*)	Species richness (0) Species diversity (0) Community composition (+) Native tree seedling % cover (+) Native vine % cover (+) Native annual % cover (+) Perennial herb % cover (0) Grass % cover (0) Shrub % cover (0)	3 years	1	9	1 m^2	1. Weeded 2. Untreated	1. 1–2% cover 2. 22% cover	Weeded in May, June, July and August for 3 years
Paul & Ledgard 2008a; New Zealand	Effect of felled pines on grassland vegetation	*Pinus nigra*	Native vegetation % cover (0) Grass % cover (+) Woody % cover (0) Native vegetation height (+) Grass vegetation height (+) Woody vegetation height (+) Native species richness (–) Grass species richness (–) Woody species richness (–)	Not reported	1	5 trees	0.5 m^2	Felled	100% killed	None
Paul & Ledgard 2009; New Zealand	Succession following wilding conifer removal	Conifers	Native vegetation % cover (56, 12, 38) Native species richness (2.9, 2.4, 7.2)	4–30 years	3	24	0.5 m^2	1. Felled <10 years ago 2. Felled >10 years ago 3. Never present	Not reported	None
		Conifers	Total vegetation cover (–) Grass cover (0) Herb cover (0) Woody species cover (–)	1 year	1	15	0.5 m^2	1. Mulched 2. Untreated	28% reduction (from 25000 stems/ha to 13000 stems/ha)	None
		Pine (*Pinus radiata*)	Native vegetation cover (100%, 5%) Native species richness (15, 3)	9 years	2	3	25 m^2	1. Stem-poisoned 2. Harvested	1. 0% cover 2. 55% cover	None

* Weed control undertaken as part of conservation management programme and not as a specifically designed removal experiment.

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AUTHOR AND COUNTRY OF STUDY	STUDY AIM	WEED SPECIES STUDIED	BIODIVERSITY VARIABLE MEASURED	LENGTH OF STUDY	NO. SITES	NO. REPLICATES	PLOT SIZE	REMOVAL TREATMENTS	LEVEL OF CONTROL ACHIEVED	FOLLOW-UP CONTROL
Pavlovic et al. 2009; USA	Impact of dame's rocket on plant community composition	Dame's rocket (*Hesperis matronalis*)	Species composition (+) Native species cover (0) Native species richness (0) Native species diversity (0) Native woody species % cover (0) Forb % cover (0) Annual % cover (0) Perennial % cover (0) Biennial % cover (0)	3 years	1	10	1 m^2	1. Hand-weeded 2. Untreated	1. <1% cover 2. c. 7% cover	Weeded spring and summer for 3 years
Reeves & Champion 2003 and Champion 2006a; New Zealand	Effect of willow removal on native plant community	Grey willow (*Salix cinerea*) Crack willow (*Salix fragilis*)	Native species cover (0) Total native vegetation cover (0) Native species richness (0)	28 months	1	6	10 m^2	1. Glyphosate sprayed aerially 2. Untreated	1. 1% cover 2. 63.3% cover	None
Rice et al. 1997; USA	Effect of herbicide treatment on structure and species diversity of native plant communities	Spotted knapweed (*Centaurea maculosa*)	Graminoid % cover and biomass: once-sprayed (+, +, +, +, +, +) resprayed (+, +, +, +, +, +) Forb % cover and biomass: once-sprayed (0, 0, 0, 0, 0, 0) resprayed (0, 0, 0, 0, 0, 0) Species richness: once-sprayed (0, 0, 0, 0, 0, 0) early picloram resprayed (–, –, –, –, –, –) all other resprayed treatments (0, 0, 0, 0, 0, 0) Plant diversity: early picloram resprayed (–, –, –, –, –, –) early clopyralid+2,4D resprayed (–, –, –, –, –, –) all other treatments (0, 0, 0, 0, 0, 0) Species composition (0, 0, 0, 0, 0, 0)	6 years	4	3 per site	9 m x 27 m	1. Early picloram 2. Late picloram 3. Early clopyralid 4. Late clopyralid 5. Early clopyralid + 2,4-D 6. Late clopyralid + 2,4-D 7. Untreated	< 10 % (biomass in sprayed plots not significantly different to untreated plots	Half of plots resprayed after 3 years

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AUTHOR AND COUNTRY OF STUDY	STUDY AIM	WEED SPECIES STUDIED	BIODIVERSITY VARIABLE MEASURED	LENGTH OF STUDY	NO. SITES	NO. REPLICATES	PLOT SIZE	REMOVAL TREATMENTS	LEVEL OF CONTROL ACHIEVED	FOLLOW-UP CONTROL
Rinella et al. 2009; USA	Evaluate native species responses to herbicide and grazing	*Euphorbia esula*	Native frequency: *Androsace* sp. (0, 0, 0, 0) *Astragalus* sp. (0, 0, 0, 0) sparse natives (0, 0, 0, 0) *Solidago mollis* (0, 0, –, 0) *Symphyotrichum falcatum* (0, 0, –, 0) *Artemisia frigida* (0, 0, –, 0) *Vicia americana* (0, 0, –, 0) *Solidago missouriensis* (–, 0, –, 0) *Achillea millefolium* (–, 0, –, 0)	16 years	1	6	12 m x 20 m	1. Grazed + aerial picloram application 2. Grazed without herbicide 3. Ungrazed + herbicide 4. Ungrazed without herbicide	No effect of herbicide on *E. esula* abundance	None
Simmons et al. 2007; USA	Effect of removal methods on weed and native community	King ranch bluestem (*Bothriochloa ischaemum*)	% cover of native species: *Bothriochloa lagaroides* (0, 0, 0, 0, 0, 0) *Solidago canadensis* (+, +, 0, 0, 0, 0)	1 year	2	5 (per treatment per site)	15 m^2	1. Early fire 2. Late fire 3. Single herbicide 4. Double herbicide 5. Single mowing 6. Double mowing 7. Untreated	Early fire ↓ 25–36% Late fire ↓ 12–23% Single herb ↑ 12–50% Double herb ↓ 12–44% Single mow ↑ 2–20% Double mow ↑ 6–10% Untreated ↑ 18–30%	For some treatments
Standish 2002; New Zealand	Compare effectiveness of hand-weeding and herbicide, and response of native vegetation	Tradescantia (*Tradescantia fluminensis*)	Native plant survival (0, 0, –, –) Native seedling abundance (0, 0, 0, 0) Native seedling species richness (0, 0, 0, 0)	20 months	1	5	5 m x 10 m	1. Hand weeded summer 2. Hand weeded winter 3. Herbicide summer 4. Herbicide winter 5. Untreated	Approx. % cover: 1. 17% 2. 6% 3. 45% 4. 7% 5. 100%	Hand weeded again and herbicide reapplied after 5–6 and 11–12 months
	Determine transplanted native seedling survival and growth with and without weeding	Tradescantia (*Tradescantia fluminensis*)	Planted native seedling survival (0) Planted native seedling growth: Karamu (0) Lowland ribbonwood (0) Mahoe (0) Long-leaved lacebark (+)	2.5 years	1	10	Circular plot 1 m in diameter	Circular plot 1 m in diameter	Not reported, but presumably similar to above	None

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AUTHOR AND COUNTRY OF STUDY	STUDY AIM	WEED SPECIES STUDIED	BIODIVERSITY VARIABLE MEASURED	LENGTH OF STUDY	NO. SITES	NO. REPLICATES	PLOT SIZE	REMOVAL TREATMENTS	LEVEL OF CONTROL ACHIEVED	FOLLOW-UP CONTROL
	Effect of weed removal methods on invertebrate community	Tradescantia *(Tradescantia fluminensis)*	Invertebrate order abundance (0, 0, 0, 0)	7 weeks	1	5	3.33 m^2	1. Hand weeded 2. Untreated	Not reported, but presumably similar to above	None
Stinson et al. 2007; USA	Impact of garlic mustard on forest understorey community	Garlic mustard (*Alliaria petiolata*)	Native species richness (0, 0) Native diversity (+, 0) Equitability (+, 0) Tree seedling abundance (0, +) Grass abundance (0, 0) Shrub abundance (0, 0) Forb abundance (0, 0)	1 year	1	4 plots of each treatment (12 plots total)	4 m^2	Hand weeded: 1. Full removal 2. 50% of original cover removed 3. untreated	1. 100% removed 2. 50% of original cover removed 3. 30–35% cover	11 months after initial control
Turner & Virtue 2006; Australia	Impact of bridal creeper on native species	Bridal creeper (*Asparagus asparagoides*)	Native species richness (0) Native grass % cover (+) Native tree % cover (0) Native saltbush % cover, Density and biomass (+)	1 year	1	4 plots of each treatment (12 plots total)	4 m^2	1. Glyphosate sponge applied by hand 2. Untreated	1. 0.1–2.4% cover 2. 35.5–44.9%	2 years after initial treatment
	Investigate plant community response following fire, with and without bridal creeper control	Bridal creeper (*Asparagus asparagoides*)	Native species richness (0) Cover of: native small shrubs (–) native creepers and climbers (–) native monocots (0) native acacias (0) other native trees and large shrubs (0)	10 years	1	15	3 m^2	1. Glyphosate sponge applied by hand 2. Untreated	1. 9.1 shoots/m^2 2. 29.7 shoots/m^2	Fire prior to herbicide application
Walker et al. 2009; New Zealand	Do weeds facilitate native plant establishment?	Scotch broom (*Cytisus scoparius*), gorse (*Ulex europaeus*) and blackberry (*Rubus fruticosus*)	Native seed germination (?, 0, –, –) Native seedling survival (?, 0, –, –) (preliminary results only)	Ongoing	1	3	Not reported	1. Sprayed 2. Roller crushed 3. Mulched 4. Root-raked 5. Untreated	3. and 4. 100% removal; remaining treatments not yet available	None reported
Webb et al. 2001; USA	Effect of Norway maple removal methods on native and exotic maple seedlings	Norway maple (*Acer platanoides*)	Native maple seedling density (0, +)	2 years	1	40	2 m^2	1. Seedlings hand weeded 2. Trees felled or girdled 3. Untreated	3 seedlings/m^2; trees not reported but assume 100% killed	None
Wester 1994; Hawai'i	Effect of weeding on threatened fern	Multiple species	Cover of threatened native fern *Marsilea villosa* (0)	3 years	1	6	1 m^2	1. Hand weeded 2. Untreated	Not reported	None

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AUTHOR AND COUNTRY OF STUDY	STUDY AIM	WEED SPECIES STUDIED	BIODIVERSITY VARIABLE MEASURED	LENGTH OF STUDY	NO. SITES	NO. REPLICATES	PLOT SIZE	REMOVAL TREATMENTS	LEVEL OF CONTROL ACHIEVED	FOLLOW-UP CONTROL
Williams et al. 1998; New Zealand	Effect of control on honeysuckle and native vegetation	Japanese honeysuckle (*Lonicera japonica*)	Native plant species % cover (0, 0, 0, 0)	1 year	1	1 (4 plots each with one spray treatment)	15 m x 8 m	Sprayed with: 1. Glyphosate 2. Triclopyr 3. Clopyralid 4. Metsulphuron	13.8% cover 27.5% cover 22.5% cover 3.8% cover	Dead honeysuckle hand-cleared in 1 m^2 subplots 6 months after spraying
	Effect of control on dock and native vegetation	Climbing dock (*Rumex sagittatus*)	Native vegetation (+) (description only)	1 year	1	16 plants	Circular plot 4 m in diameter	Sprayed with metsulphuron	15 of 16 plants killed	None
	Effect of control on bone-seed and native vegetation	Bone-seed (*Chrysanthemoides monilifera* ssp. *monilifera*)	Native plant species % cover (0)	15 months	1	2 transects 12–14 plots	30 m transects, 1 m^2 plots	Sprayed with triclopyr	19–36% cover	None
Yanez et al. 2004; Ecuador*	Effect of weed control and planting on restoration	Multiple weed species incl. blackberry (*Rubus niveus*) and guava (*Psidium guajava*)	Native species % cover (+)	3 years	1	10	5 m^2	1. Chemical spray (picloram and metsulfuron for guava and glyphosate for blackberry) 2. Untreated	Not reported	None

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