Fire, Weeds and the Native Vegetation of New South Wales

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1. Background to the review

This literature review forms part of a suite of materials that constitute the technical foundation of the Hotspots Fire Project. Previous literature reviews have been prepared for specific Catchment Management Authority regions (now Local Land Services), reviewing and reporting on the role of fire within native vegetation in each region. This review has been prepared at the statewide scale and is intended to review and report on the interactions of fire with weeds that occur in the native vegetation of New South Wales. While most Hotspots products are targeted at landholders, literature reviews are directed towards a professional audience. Their primary aim is to provide the ecological background to underpin and inform the messages about fire (and in this case fire and weeds) that Hotspots and local natural resource management (NRM) practitioners present at various workshops and field days. A secondary aim is to offer a platform for discussion and debate on the role that fire plays in managing weeds as well as the role fire plays in weed establishment and spread in the native vegetation of New South Wales. In both cases, we hope the outcome will be more informed management of fire, weeds and natural ecosystems.

This review considers literature relevant to the interactions of fire with weeds in all vegetation communities across New South Wales, with a specific focus on a limited subset of significant weeds. These weeds are either Weeds of National Significance, are known to significantly affect native vegetation communities, or are weeds for which fire is understood to play a major role in management and control. It is designed to be used in tandem with regionally specific literature reviews of fire and vegetation covering the Northern Rivers, Southern Rivers, Central West; Hunter Central Rivers, Namoi; Lachlan, Border Rivers–Gwydir, Murray and Murrumbidgee catchment management authority (CMA) regions (Watson 2006 a, b, c, 2007; Tierney & Watson, 2009 a, b; Graham, Watson & Tierney 2013, 2014, 2015 a, b). These are available via the Hotspots website: www.hotspotsfireproject.org.au. The Hawkesbury–Nepean CMA area is covered in Vegetation, Fire and Climate Change in the Greater Blue Mountains World Heritage Area (Hammill & Tasker 2010).

To our knowledge there has been no other review undertaken which has examined the interaction between fire and weeds in native vegetation across the State. This knowledge gap has been identified in the process of collating and assessing the available information in response to an ongoing interest expressed by participants at Hotspots Fire Project workshops.

The intent of this resource is to provide a platform for further discussion and for the document and content within to continue to expand and evolve as our understanding of the complex interaction between fire and weeds in native vegetation across the State increases**.**

Defining weeds

Generally speaking, weeds are plants that have been introduced to a place; they are 'exotic or alien species' (Michael 1981), although in certain instances this may include Australian native species introduced outside their native range. There are many definitions of a 'weed', largely dependent on the perspective from which the plant is viewed (e.g. agricultural, ecological or legislative) and the intended management of it. The simplest definition of a weed is 'a plant growing where it is not wanted'. For the purposes of the *Australian Weed*s *Strategy* (DEWR 2006): 'a weed is considered pragmatically as a plant that requires some form of action to reduce its harmful effects on the economy, the environment, human health and amenity'.

Categories, classifications, types and legal status of weeds

Within New South Wales and nationally, weeds fall under a number of categories and classification systems, detailed within the following sections.

The legal framework for weeds in New South Wales

The *Noxious Weeds Act* has been replaced by the *Biosecurity Act 2015*. The broad objectives for biosecurity in New South Wales are to manage risks from animal and plant pests and diseases, weeds and contaminants by:

- preventing their entry into New South Wales
- quickly finding, containing and eradicating any new entries
- effectively minimising the impacts of those pests, diseases, weeds and contaminants that cannot be eradicated through asset based protection.

Specifically, the *Biosecurity Act* will:

- embed the principle that biosecurity is a shared responsibility
- provide modern, flexible tools and powers that allow effective management of pests and diseases, weeds and contaminants across the landscape regardless of whether it is private or public land

- minimise delays and define responsibilities in emergency situations
- provide for risk-based decision-making that enables a flexible approach to responding and managing biosecurity risks regardless of the type of biosecurity matter
- support a national approach to biosecurity and give effect to intergovernmental biosecurity agreements.

Weed management actions under the Act will be implemented under eleven regional plans and focus on four main categories: - Prevention, Eradication, Containment and Asset Based Protection (NSW DPI, 2013). Asset based protection includes economic, social or environmental assets. Weeds not listed must still be considered under the following provision:

"All plants are regulated with a general biosecurity duty to prevent, eliminate or minimise any biosecurity risk they may pose. Any person who deals with any plant, who knows (or ought to know) of any biosecurity risk, has a duty to ensure the risk is prevented, eliminated or minimised, so far as is reasonably practicable."
https://www.lls.nsw.gov.au/biosecurity/weed-control

Australian weeds strategy

Weeds of National Significance

Weeds of National Significance (WONS) are regarded as some of the worst weeds in Australia because of their invasiveness, potential for spread, and economic and environmental impacts. Developed as a key outcome of the *national weeds strategy* (ARMCA 1997) the original WONS list, endorsed in 1999 included 20 species. A further 12 species were added in 2012. This review considers seven WONS species because of their occurrence within, and significant impact on fire-dependent and adjacent environmental and cultural assets in areas of New South Wales. These are:

- bitou bush (*Chrysanthemoides. monilifera* subsp. *rotundata*) / boneseed (*C. monilifera* subsp. *monilifera*)
- blackberry (*Rubus fruticosus*) species aggregate
- Chilean needle grass (*Nassella neesiana*)
- lantana (*Lantana camara*)
- Scotch broom (*Cytisus scoparius* subsp. *scoparius*)
- serrated tussock (*Nassella trichotoma*).

National Environmental Alert List

Twenty-eight environmental weeds were identified as National Environmental Alert Weeds. They are non-native plant species that are in the early stages of establishment and have the potential to become a significant threat to biodiversity if they are not managed.

Native plants considered weeds

Australia is a large nation with a globally significant and diverse array of biomes, within which there are over 21,000 native vascular plant species known (Chapman 2009). Some Australian native plants have become invasive in areas beyond their natural range or habitat. This is attributable to several factors including their popularity as ornamental plants, and inappropriate use of species or genotypes in restoration projects. Specific examples include use of horsetail she-oak (*Casuarina equisetifolia*) and coastal tea tree (*Leptospermum laevigatum*) in regeneration programs following strip mining of mineral sands along the NSW coast in areas where they did not naturally occur. Several inappropriate wattles (*Acacia* spp.) have been used in 'spray mulch' following road construction (M. Graham 2000–2017, pers. obs.). Landscape, climatic changes and altered fire regimes may have resulted in native species such as sweet pittosporum (*Pittosporum undulatum*) becoming invasive due to mesic shifts and nutrient enrichment (where conditions have changed in favour of moisture-dependent species over those that are drought tolerant), particularly in sandstone landforms (see Mullet 1999 and Rose & Fairweather 1997).

This review will not give further consideration to Australian native plant species as weeds.

Scale and scope of the review

This document aims to review literature relevant to the interactions of fire with weeds in the native vegetation of New South Wales. In particular, the review will consider widespread weeds that are known to significantly degrade native vegetation.

There are 8934 vascular plant species or subspecies recorded within New South Wales, of which 1878 (21%) are listed as 'introduced' (Royal Botanic Gardens Trust 2016). Many introduced species occur in isolated pockets of the State or exist as very small populations. There is very little published literature, data or expert opinion available for these species so they will not be considered by this review. Neither will agricultural weeds or non-native ecosystems be considered by this review.

In addition to the seven WONS listed above, the following weeds are to be considered by this review, based on discussions with participants of Hotspots Fire Project workshops, project partners, bush regenerators and fire management practitioners: -

- African lovegrass (*Eragrostis curvula*)
- African olive (*Olea europaea* subsp. *cuspidata*)
- camphor laurel (*Cinnamomum camphora*)
- Coolatai grass (*Hyparrhenia hirta*)
- large-leaved privet (*Ligustrum lucidum*) and small-leaved privet (*L. sinense*)
- phalaris grass (*Phalaris aquatica*)
- South African pigeon grass or setaria (*Setaria sphacelata* var. *sericea*)

Caveats

This review does not address native species regarded as weeds, nor does it cover most of the introduced species known to occur within New South Wales because of a lack of data. Instead, it focuses on widespread weeds that are known to significantly degrade native vegetation and for which there is published literature, 'grey literature', unpublished data and expert opinion available regarding the interactions of fire with those weed species and the native vegetation of New South Wales.

New South Wales is an incredibly biodiverse state with a rich continuum of vegetation formations, subformations, classes and communities. Within each, many dynamic and varied processes are in operation, such as variations in climate and fire regime. These strongly influence ecosystem dynamics and in turn influence the vulnerability of a particular ecosystem to degradation by weeds, and determine the potential for a particular weed to invade or degrade that ecosystem. The presence of weed propagules within a particular landscape is the key factor that determines whether a particular ecosystem, or fragment of it, is vulnerable to degradation by that weed.

In summary, every site in New South Wales is different; all are subjected to different dynamics and interplays of natural and anthropogenic processes. The same weed species may behave completely differently across multiple sites and within different ecosystems. As a consequence, broad-reaching conclusions about the influence of fire on a particular weed species within a vegetation community are difficult and in most instances impossible.

2. Summary of key findings and principles

The following key concepts and principles were identified in a review of a wide range of literature sources concerning the interaction of fire with a selection of environmental weeds in New South Wales native vegetation communities:

- There is a general lack of specific knowledge of the interaction of fire with most of the weeds that are degrading the native vegetation of New South Wales. This is preventing the achievement of good weed and fire management outcomes.

- The interactions of fire with weeds are complex, varied and often difficult to predict. A thorough knowledge of the ecology and vital attributes of a weed and the ecosystem in which it is present is needed if fire is to be used as a management tool.

- Land managers using fire for any purpose should include resourcing for weed control measures as part of routine planning.

- Fire can be useful for providing access and reducing the mass of a target weed to enable more effective mechanical or chemical treatment.

- Unplanned wildfires can present an opportunity for opportunistic weed control actions.

- Many of the weeds that impact shrubby Wet Sclerophyll Forests are fleshy fruited species that are dispersed by birds and flying-foxes from distant sources. There is limited value in using fire to deplete weed seeds in the soil seed bank but it may be useful as a 'once off' to flush out long-lived weed seeds for herbicide or manual control and promote native species germination.

- In some sites, especially in northern New South Wales where dense lantana is present, infrequent fire may be useful as part of an integrated strategy with mechanical or herbicide treatment to kill mature plants. Site conditions will dictate the order of treatment as moisture will limit the ability to introduce fire into many areas at most times of the year. It may be necessary to kill mature lantana plants with herbicide first in order to have enough dry fuel for combustion.

- In sites with South African pigeon grass *(Setaria sphacelata* var. *sericea)* (or several other weeds) present, fire should not be used in isolation as it is likely to promote growth and exacerbate existing infestations. For this weed and most of the others reviewed by this report, combinations of treatments that include fire and herbicide application will achieve the best outcomes.

3. The impacts of weeds in New South Wales

Globally, weeds generate numerous significant negative impacts (Richardson *et al.* 2000; Tilman & Lehman 2001), including being the second-most serious cause of ecological degradation and loss of biodiversity (Sala *et al.* 2000; Secretariat of the Convention on Biological Diversity 2006), as well as causing major economic and social costs (Sinden *et al.* 2004).

Economic costs

Weeds cause a reduction in productive output from the rural industries of New South Wales. This constitutes both a direct reduction in productivity and an economic cost to undertake weed control. The cost of managing weeds in NSW is almost $2 billion annually without accounting for social and environmental costs (NRC. 2014). The cost to the environment is difficult to estimate due to a lack of reliable data, but it would be similar to, or greater than, the cost to agricultural industries (Sinden *et al.* 2004). There is a need for the economic costs of weed invasion on natural ecosystems to be better studied and quantified so the true costs can be established and to enable better-informed management and prioritisation of weeds (Hobbs & Humphries 1995; Byers *et al.* 2002).

Impacts on biodiversity

Almost all native vegetation communities within the State have been invaded by weeds, or are vulnerable to invasion (Coutts-Smith & Downey 2006; Downey 2008; Williams *et al.* 2009). The exceptions are generally nutrient-deficient ecosystems such as coastal sandplains composed of highly infertile siliceous sandy soils (Keith 2004), but these habitats can be invaded by a limited subset of weeds, some of which are potentially ecosystem transformers.

Significant weed invasions change the diversity and structure of native plant communities (Carr, Yugovic & Robinson 1992; Downey & Leys 2004; Grice, Field & McFadyen 2004; McDougall & Turkington 2005), causing major shifts in the composition and availability of resources such as nectar and pollen (Potts *et al.* 2010), grasses and other seed sources (McArdle, Madolny & Sinden 2004), fleshy fruits (Neilan *et al.* 2006; Kanowski, Catterall & Neilan 2008), foliage, leaf litter, habitat structures such as hollows and nutrient availability. Weed invasions can also cause significant shifts in the prevailing fire regime at a particular site (Brooks, Fonseca & Rodrigues 2004; Aires, Bell & Matthews 2013; Aires 2014).

Within New South Wales and Australia, weeds are one of the greatest threats to biodiversity (Groves 1986; Coutts-Smith & Downey 2006; Commonwealth of Australia 2009) and the natural environment, ranked second behind the clearance and fragmentation of native vegetation (Adair & Groves 1998; Coutts-Smith & Downey 2006). Major weed invasions change the structure and function of ecosystems (Downey & Leys 2004). These changes impact adversely on many native plants and animals because weeds compete with native plants for space, nutrients and sunlight (Carr, Yugovic & Robinson 1992; Downey & Leys 2004; Grice, Field & McFadyen 2004; McDougall & Turkington 2005).

The severity of the threat of weeds relative to other threatening processes

In a first for Australia, Coutts-Smith and Downey (2006) analysed the impacts of weeds on the threatened species and ecological communities of the State. They found that weeds posed a threat to 45% of the biodiversity examined. A key finding was that, as a single factor, the threat posed by weeds was ranked second after land clearing, posed a similar threat to that of altered fire regimes (with which weed invasion synergises), and was greater than the threat posed by pest animals. Weeds were also ranked highly when compared with other major threatening processes such as the destruction and modification of native vegetation.

Impacts on threatened species, populations and ecological communities in New South Wales

Coutts-Smith and Downey (2006) found that weeds threatened 419 listed threatened entities, comprising:

- 279 threatened plants (166 endangered and 113 vulnerable)
- 62 animals (30 endangered and 32 vulnerable)
- 14 endangered populations
- 64 endangered ecological communities.

The findings of Coutts-Smith and Downey (2006) indicate that almost half of all the listed species and communities in New South Wales are threatened by weeds.

Weeds posing a threat to multiple threatened species

Coutts-Smith and Downey (2006) identified 127 individual weed species (from 120 genera and 51 families) as threatening 204 threatened entities. For the remaining 215 entities threatened by weeds, a specific weed species could not be identified; rather the threat was described as ‘weed invasion’, or the threat was a particular weed genus (e.g. Salix). According to Coutts-Smith and Downey (2006) the five weed species most commonly identified as threatening biodiversity in New South Wales include:

- lantana (96 threatened entities)
- bitou bush (46)
- blackberry (21)
- kikuyu (*Pennisetum clandestinum*) (16)
- Scotch broom (12).

4. Weeds and disturbance

Many weeds require disturbance in order to establish (Hobbs & Hunneke 1992) and weeds frequently thrive and reproduce prolifically in disturbed environments. Weeds are often the first species to colonise and dominate sites following disturbance including disturbance caused by fire (Hobbs & Hunneke 1992; Ross, Fox & Fox 2002). Most disturbances to native vegetation communities that result in weed invasion are of an anthropogenic nature (including clearing, slashing, cultivation and herbicide use). However, fire is a process that can be instigated by either anthropogenic or natural agency and can create conditions conducive to weed invasion, particularly where other processes such as nutrient enrichment are operating (Thomson & Leishman 2005).

Following disturbance, weeds are often able to out-compete native species because they are not exposed to the pressures of pests, diseases and predation that they are subjected to in their original native habitats (Keane & Crawley 2002). In many instances, disturbance provides resources and conditions that facilitate the establishment and rapid growth of invading weed species to the detriment of native species. The availability of mechanically disturbed and exposed mineral soil, an ash bed following fire, or layers of decomposing organic matter following herbicide application are all examples of conditions that enable the establishment of weeds and which may preferentially advantage weeds over native species in many Australian landscapes (DEWR, 2006). This can lead to significant weed populations post-disturbance, a decline in ecosystem integrity and health, and a reduction in resources such as nectar, foliage and seeds (all key foraging resources for native animals) in the post-disturbance environment. This then contributes to further species extinctions from direct competition or through the loss of resources and ecological niches (Adair & Groves 1998).

In post-disturbance regeneration, weeds often grow faster than native species, out-competing the original flora for access to space, water, sunlight and nutrients (Blossey & Notzold 1995). If this process continues through repeat disturbance events, the structure of the original native vegetation can be significantly altered and the diversity of native species significantly reduced. As these trends proceed over time thresholds are crossed beyond which recovery of the original structure and floristics is not possible and a permanently altered state or trajectory results (SER 2004).

Weeds and nutrient enrichment

Many landscapes across the coastal fringe of New South Wales and within agricultural areas have modified nutrient regimes because of the existence of nearby and adjoining urban and agricultural developments (Clements 1983; Leishman 1990; King & Buckney 2002). In many instances elevated nutrients and toxins in run-off from residential or agricultural developments has resulted in weed invasions into previously intact native vegetation communities (Allcock 2002; Riley & Banks 1996; Leishman, Hughes & Gore 2004) and resulted in highly modified fire regimes (Lake & Leishman 2004). Mesic shifts in vegetation communities have been widespread as a result of the invasion of previously drier native vegetation communities by densely crowned, fleshy fruited weed species that are dispersed by birds such as the pied currawong (*Strepera graculina*) (Buchanan 1989). Mesic shifts can contribute to declines in the resilience and health of native vegetation communities (Thomson & Leishman 2004), and it is likely that some of these weed invasions or the prior nutrient enrichment of the environment have resulted in permanent and irreversible degradation and shifts to an entirely new nutrient status, vegetation community (King & Buckney 2001) and fire regime (Leishman & Thompson 2005).

5. Fire ecology general principles

A brief summary of some of the basic principles regarding fire in the Australian landscape is provided below. A more in-depth examination of these concepts can be found in section 1 of *Fire and the vegetation of the Murray Valley* (Graham, Watson & Tierney 2015a).

Fire in the landscape

In simple terms, fire is the chemical reaction which occurs during the rapid oxidisation or combustion of <u>fuel</u>. This requires an ignition source to begin and a constant supply of <u>oxygen</u> and <u>heat</u> to sustain the chemical reaction. Fire will not take place if one of these three elements is significantly reduced or removed. The combustion of fuel releases carbon dioxide, water and energy in the form of heat and light as the bonds holding the biomass together are broken and rearranged. In a bushland environment, the fuel is the vegetative biomass made up of trees, shrubs, vines, grasses, forbs and peat. The climate and terrain or landform influences how much oxygen and heat are available on any given day and over longer time periods. 'Wildfire' is that which is unplanned or is burning uncontrolled in areas of combustible vegetation or soils such as grasslands, bushland or peat environments. A planned and lawful fire is generally referred to as a 'prescribed fire' as it is lit under particular controlled conditions to meet specific objectives and management actions.

Succession and disturbance

The concept that plant communities existed in a particular state of succession on the way to a climax community was developed by early ecologists such as Clements (1916, 1936). Clements argued that a vegetation community is an organic entity which is born after disturbance, grows, matures and dies. An area of moss and lichen will eventually become a climax forest through a series of successional stages or seral units. The presence of plants at a certain stage of succession always leads to plants from the next seral stage as they change the environment not to suit their own propagation but that of the next seral stage. If a disturbance event happens the process is reset but follows the same predictable path dictated by the climate of the continent or area. This has been referred to as the equilibrium model or paradigm and the climax state was viewed as being in balance and as a desirable conservation goal (Wu & Loucks 1995).

Noble and Slatyer (1980) found that this concept of equilibrium and succession did not fit vegetation communities which had regular disturbance events, such as fire. Their research found that the species present at a site at the time of disturbance had a major influence on how the community will develop until the next disturbance event. They presented a model based on the vital attributes of species at a particular site in order to better understand the dominance and dynamics of the species present (Noble & Slatyer 1981). Westoby, Walker & Noy-Meir (1989) presented a model that stated that systems could exist in different states at different times or within the same community depending on disturbance events and ongoing pressures. This non-equilibrium model is now recognised as being more representative of the different patches or state of flux that Wu and Loucks (1995) observed in vegetation communities and that a range of states is needed to conserve biodiversity and restore ecosystem function (Clewell & Aronson 2013).

Fire regimes

No two fires are the same in how they impact on a particular site. The behaviour of a fire event is determined by the interaction of the fuel, terrain and weather. It is also dependent on the cumulative effects of previous fire events or conversely, the lack of fire over time (Cary 2002). The history of fire at a particular site is known as the fire regime. A fire regime at a given site is the combination of the type, frequency, extent, intensity and seasonality of fire events (Gill 1975). The fire type is whether it is burnt above or below the ground such as in a peat fire. The frequency is the interval between fires. The fire extent is the degree of patchiness across an area. All of the components combine to influence how fire behaves on a given day, but also how vegetation responds after a disturbance event and what niches are available for biota to persist in. Residence time is another attribute which is difficult to measure. This is the time an area was exposed to heat and smoke during a fire. This will influence the germination or death of seeds in the soil seed bank. This is significant for managing weeds at a site. Scientists and land managers alike now realise they must understand the many variations in fire regimes over long periods in a range of areas to better understand the effects on biodiversity (Bradstock, Williams & Gill 2002; Whelan *et al.* 2006).

Species responses to fire

How different plant species respond to fire depends on their life cycle, the fire regime at a given site and the post-fire environment. To understand how weeds may compete with native species at a site, land managers must consider these variables. Gill (1981) details a range of important aspects of plant responses but also provides a simple method of classifying plants as ‘non-sprouters’ or ‘sprouters’.

Non-sprouters

Non-spouters are mature plants which die when subject to 100% leaf scorch by fire. For these species to persist at a site they must reproduce from seeds which can survive or avoid the effects of fire. The seeds are stored on the plant, in the soil or are brought in from an unburnt area. Plants which can only reproduce from seed after fire are referred to as obligate seeders. These plants can be vulnerable to extinction in a location if the sources of seed are depleted and successive fires occur before they reach reproductive maturity and replenish the supply of available seed. These plants are considered fire sensitive even though a single fire event may not lead to extinction, but a regime of frequent fire will. These species often have fast growth rates to reach maturity quickly and take advantage of the light, space and nutrient availability after a fire. Seed longevity in the soil is a common feature with these species (Noble & Slatyer 1981).

Sprouters

These are plants which in the same situation survive by re-sprouting from regenerative buds protected underground or beneath layers of bark. They do this from root suckers and basal stem sprouts or from epicormic buds and other above-ground growth nodes (Gill 1981). Many of these species are tuned to reproduce only after fire so they will be disadvantaged in a regime of low fire frequency. Seeds in the soil lose viability and less fire-sensitive species may become dominant (Noble & Slatyer 1981).

Thresholds

How species respond and adapt to fire regimes rather than just a single event has become the subject of research and ultimately management in recent years (Bradstock, Williams & Gill 2002). The non-equilibrium model using the vital attributes approach of Noble and Slatyer (1980) has been used to develop fire management guidelines for broad vegetation types in New South Wales (Kenny *et al.* 2003). Species in the broad vegetation types (Keith 2004) are grouped together based on their vital attributes. These are selected in regards to the species sensitivity to fire or the method of reproduction after fire (Kenny *et al.* 2003).

Information on the vital attributes of plants needed to set the minimum thresholds is relatively easy to obtain for many plants, so the lower thresholds have a greater degree of confidence. The data required for seed and adult longevity is much more difficult to obtain, so the upper thresholds may be based on less robust data (Watson 2006a).

6. Principles of ecological restoration

What is ecological restoration?

‘Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed. It is an intentional activity that initiates or accelerates an ecological pathway — or trajectory through time — towards a reference state’ (Gann & Lamb 2006).

A reference state is an example of an actual or historical ecosystem which is well defined. The progress of a restoration project can be measured against this reference ecosystem. In many ecosystems, however, it may not be possible to define the former state as this may not be known and present conditions and constraints may prevent it ever being restored to the same state. Instead, an ecological trajectory which is moving towards this reference state, as best as it can be defined, should assist ecosystem health and functional integrity. The reference state should be viewed as dynamic and while it should be based on existing and historical ecosystems as much as possible it must also be sustainable. The objectives of an ecological restoration project should define a clear ecological pathway or trajectory along which management actions are moving the ecosystem towards the desired reference state (Clewell & Aronson 2013).

Gann and Lamb (2006) detail a series of guiding principles which good ecological restoration practice should consider:

- incorporating biological and environmental spatial variation into the design
- allowing for linkages within the larger landscape
- emphasising process repair over structural replacement
- allowing sufficient time for self-generating processes to resume
- treating the causes rather than the symptoms of degradation
- include monitoring protocols to allow for adaptive management.

The document *National standards for the practice of ecological restoration in Australia* (SERA 2016) provides a comprehensive guide for designing and undertaking restoration projects in Australia.

Fire and ecological restoration

Bush regeneration in Australia has traditionally sought to restore or establish a functioning ecological system which, in an ideal world, is ultimately self-sustaining. Early bush regeneration techniques such as the Bradley method were developed in the 1960s and 1970s. The guiding principles are to secure the least disturbed areas first, minimise further disturbance and avoid treating new areas faster than the rate of natural regeneration (Bradley 2002). These basic principles are still followed by many bush regenerators today, but in areas of severe degradation with a growing array of voracious weeds other approaches are also needed when significant conservation or cultural heritage assets are threatened.

The use of more disruptive methods such as large teams, herbicide, machinery and fire is now being used in areas where urgent action is required to prevent the decline or loss of a significant population of a critically endangered species such as the northern population of the eastern bristlebird (*Dasyornis brachypterus)* in the Border Ranges area (Parker, 2014). These methods are also required in severely degraded ecosystems where the natural regenerative processes are not keeping pace with 'transformer weeds' (Richardson *et al.* 2000). Transformer weeds are weeds that, if left unchecked, can dominate an area and effectively neutralise the natural regenerative processes and functioning of the ecosystem.

Fire has had a major influence on the evolution of vegetation and many of the ecosystems found in Australia (Bowman *et al.* 2012). It can be argued that fire regimes should be considered as a major element of any land management planning in most, if not all, vegetation types. Ecological restoration and weed management planning need to consider the effects of different fire regimes on a site; this may include fire exclusion in some areas. Fire management planning must also consider the effects of fire regimes and fire events on the abundance of weed populations (Williams 2008). A particular fire regime may promote specific weeds which may in turn increase fuel loads.

The presence of some weeds may prevent the implementation of a fire regime which is sympathetic to the sustainability and functionality of the ecosystem, or may prevent hazard reduction burning needed to reduce fuel hazard loads which are a risk to life and property. DiTomaso and Johnson (2006) examined the available knowledge on the use of fire to control invasive plants in wildland areas in the United States of America (USA). They found that fire is used to control weeds 'through direct damage and suppression of the target species, or as part of an integrated approach in which fire facilitates more effective use of another control strategy, including mechanical, cultural, or chemical options' (DiTomaso & Johnson 2006). A comparable review has not been undertaken in Australia.

There can be advantages and disadvantages when using any weed control technique. Muyt (2001) detailed this for a range of commonly used weed control methodologies. In terms of fire (e.g. controlled burn, spot burns, pile burns), Muyt listed a number of advantages or benefits and disadvantages or limitations as follows.

Advantages:

- selective (spot burns)
- removes excess foliage (for follow-up treatments)
- supplements other methods
- minimises risks to indigenous flora regeneration
- encourages germination of soil-stored weed seed bank
- inexpensive.

Disadvantages:

- non-selective
- usually does not eradicate weeds
- inappropriate for non-fire adapted areas
- seasonal and timing limitations
- encourages weed growth/germination
- altered nutrient/moisture availability can favour weeds
- potential for run-off/erosion
- fauna, people and property risks
- specialist knowledge required.

In fire dependent vegetation, where a decision has been made to use fire as a management tool, flexible fire regimes are needed which may allow the use of frequent fire followed by weed control actions to restore the systems trajectory towards a desired state. In this way, fire can be used as a direct mechanism to remove or control the invading species and allow native species to compete and replace them. A thorough knowledge of the life attributes of weeds and native species and their fire responses is needed to achieve the best management outcomes.

There are many ways to incorporate the direct use of fire into ecological restoration. Fire can be used to remove the biomass of a weed which can be restricting the growth of native species (Lindenmayer *et al*.2015; Thomas *et al*. 2006). The invasion of perennial grasses into grassy woodlands and native grasslands is an example of this. Fire can remove competing weeds and stimulate native vegetation which can shade out weeds in some vegetation types or contribute seeds to the soil seed bank to improve resilience (Lindenmayer *et al*.2015; Thomas *et al*. 2006).

Fire can kill the seeds of invasive species stored in the soil. It can also stimulate the germination of weed seeds in the soil which are then flushed out and can be removed by other means, for example mechanical or herbicide (Thomas *et al*. 2006). This may increase the amount of weed present at a site in the short term but can deplete the soil seed bank of the weed which can be replaced by native species and move the ecosystem along an ecological trajectory towards a healthier state. Hand sowing of native species may be required. Though this is more labour intensive and resource-hungry in the short term, it may mean that the ecosystem is more resilient to the impacts of disturbance in the future (Firn *et al.* 2008). This would be particularly relevant in remnant native grasslands and areas affected by various forms of Eucalypt dieback where conservation or cultural values are present or adjacent more intact areas may become threatened if the problem persists.

7. Vegetation formations in relation to weeds and broad environmental gradients

The native vegetation communities of New South Wales span a number of very broad physical gradients that cover substantial variations in:

- mean average annual rainfall from east to west — from greater than 2000 mm atop the wettest mountains on the North Coast to less than 100 mm west of Broken Hill) (BoM 2011)
- temperature — from the subtropical climate of the North Coast to the alpine climate of the Australian Alps
- altitude — from sea level to 2228 m above sea level (ASL) at Mt Kosciuszko
- soil type — from pure siliceous sands on the coastal plain to fertile clay-loam and cracking soils derived from basalt on the Monaro and Liverpool Plains and Dorrigo Plateau and water-logged organic peaty soils in the montane wetlands atop the Great Dividing Range.

This variation has a profound influence on both the type of native vegetation occurring and the range of weeds present or able to occupy a site. The interactions between the plants and animals as well as the past disturbance history (e.g. fire, storm, drought and human modification) all influence the vegetation community found at a site. These interactions can vary over time and at different situations along the environmental gradients. The range of interactions is complex but general patterns can be inferred for identifying broad vegetation formations (Keith 2004).

Water

Water has a major influence on vegetation and is probably the limiting factor in most situations. Sources can be from dew, mist and snow as well as rain. Dams, irrigation and groundwater depletion may also influence the vegetation in agricultural and urban areas. Water availability over time influences what vegetation persists. This can vary locally with altitude and from north to south with latitude. As the amount and reliability of rainfall decreases from the coast to western New South Wales, moisture-loving vegetation such as rainforests disappears. In the subtropical areas of northern New South Wales most rain falls in summer and autumn, but on the South Coast more rain falls in winter (Keith 2004). This not only has a major influence on how species grow and persist, it also has a major influence on the frequency and intensity of wildfires. On the south coast the peak fire season is at the hottest and driest time of the year; summer, in contrast to the north coast which frequently experiences hot wildfires at its driest time in spring (Sullivan *et al.* 2012).

Soils

The texture and mineral make up of soils provide the medium in which plants grow and nutrients for vegetation to metabolise. The amount of water available in the soil influences how the nutrients can be taken up through plant roots and redistributed by soil microbes. Fertile soils with high rainfall will support tall forests, but if rainfall is limited then grassy woodlands and grasslands may develop. On low fertility soils such as coastal dunes and sandstone areas the vegetation grows slowly and has sclerophyllous leaves and other adaptations to survive periods of drought (Keith 2004).

The vegetation formations found in these dry sclerophyll communities usually contain high numbers of resprouters (Gill 1981) and other adaptations to living with fire (Bowman *et al.* 2012).

Temperature

Temperature influences how plants can exploit the water, light and nutrients at a given site. Some species grow well in cooler conditions, but as average or maximum temperatures increase along gradients, growth or reproduction may be inhibited. Plants in warmer environments are also more susceptible to predation due to changes in their chemical structure. Many cool temperature species have mechanisms which make them frost hardy or shut down during winter. Average temperatures decline with increasing altitude and from north to south. The diversity of rainforests declines with temperature from the north to the south of the State. In the extreme cold of alpine areas only a small group of highly specialised plants can persist (Keith 2004).

There are many other physical factors which also influence how different plants persist. The interactions of these factors all vary along gradients. Invasive plants may have special features which enable them to better exploit these factors along gradients and take advantage of the conditions available after disturbance.

Kenny *et al.* (2003) provide fire threshold guidelines which take into account many of these factors in relation to the appropriate use of fire in vegetation formations for New South Wales. While these thresholds are broad and may not provide precise information for using fire as a management tool to control or remove weeds, the recommended fire intervals provide a starting point for planning a fire regime suitable to restore the desired vegetation formation at a site.

Knowledge of the vital attributes and fire response data (Noble & Slatyer 1981; Gill 1981) for plant and weed species at a site will greatly enhance the success of using fire as a tool for ecological restoration.

Fire and weeds in Rainforests

Rainforests are forests of broad-leaved mesomorphic trees, with vines, ferns and palms found on the coast and tablelands in mesic sites on fertile soils. The main plant families include Cunoniaceae, Sapindaceae, Monimiaceae, Apocynaceae and Rubiaceae (Keith 2002).

<u>Fire as a restoration tool</u>

Rainforests can survive occasional fire but are vulnerable to frequent fire, so fire is generally excluded (Kenny *et al.* 2003). Although subject to a wide range of invasive weeds it is not applicable to use fire as a control measure in rainforests. A wildfire event may provide the opportunity to access areas of untreated weed infestations and undertake follow-up seedling control.

Fire and weeds in Wet Sclerophyll Forests

Wet Sclerophyll Forests shrubby subformation

Shrubby Wet Sclerophyll Forests are tall forests of scleromorphic trees (typically eucalypts) with dense understoreys of mesomorphic shrubs, ferns and forbs. They are found in relatively fertile soils in high rainfall parts of coast and tablelands. The main plant families are Myrtaceae, Rubiaceae, Cunoniaceae, Dryopteridaceae, Blechnaceae and Asteraceae (Keith 2002).

Fire as a restoration tool

Kenny *et al.* (2003) recommend a fire interval of between 25 and 60 years; and avoiding crown fire in the lower end of the interval range due to the presence of many fire-sensitive rainforest elements which would be replaced with fire-tolerant sclerophyllous species. Shrubby Wet Sclerophyll Forests are particularly susceptible to weed invasion following disturbance such as fire because they have moderate to high soil fertility and soil moisture. The invasion of aggressive transformer weeds such as lantana, camphor laurel, and large-leaved privet within Wet Sclerophyll Forests across south-east Australia has been well documented. For example Gooden *et al.* (2009) found that when lantana reached cover abundance values above 75% it had a dramatic effect on all major structural groups. This resulted in significant species loss occurring, particularly of trees and shrubs which resulted in a shift from tall open forest to a low, dense lantana-dominated shrubland.

Many of the weeds that impact shrubby Wet Sclerophyll Forests are fleshy fruited species that are dispersed by birds and flying-foxes from distant sources. There is limited value in using fire to deplete weed seeds in the soil seed bank but it may be useful as a 'once off' to flush out long-lived weed seeds for herbicide or manual control and promote native species germination. In some sites, especially in northern New South Wales where dense lantana is present, infrequent fire may be useful as part of an integrated strategy with mechanical or herbicide treatment to kill mature plants. Site conditions will dictate the order of treatment as moisture will limit the ability to introduce fire into many areas at most times of the year. It may be necessary to kill mature lantana plants with herbicide first in order to have enough dry fuel for combustion.

Wet Sclerophyll Forest Grassy subformation

Grassy Wet Sclerophyll Forests are tall forests of scleromorphic trees (typically eucalypts) with grassy understoreys and sparse strata of mesomorphic shrubs. They are found on the coast and tablelands in high rainfall regions and along major inland watercourses on relatively fertile soils. The main plant families include Myrtaceae, Poaceae, Euphorbiaceae, Fabaceae, Casuarinaceae and Asteraceae (Keith 2002).

Fire as a restoration tool

Kenny *et al.* (2003) recommend a fire interval of between 10 and 50 years; avoiding crown fire in the lower end of the interval range; and having some intervals greater than 15 years. The lower threshold reflected the dominance of grassy species in the understorey and high presence of resprouters compared to obligate seeders. Grassy Wet Sclerophyll Forests occupy drier, more exposed and elevated sites than shrubby Wet Sclerophyll Forests. With more frequent fire this subformation generally has a greater proportion of grasses in the understorey, but with less frequent fire shrubs and weeds can dominate the understorey. With lower moisture levels, shallower and less fertile soils grassy Wet Sclerophyll Forests are less prone to invasion by the many mesic weeds known from Rainforests and shrubby Wet Sclerophyll Forests, but in long periods without fire lantana can form dense infestations.

Grassy Wet Sclerophyll Forests, particularly those at the driest end of the spectrum are vulnerable to invasion and degradation by a range of exotic grasses and wind-dispersed herbs and forbs, whilst those at the wetter end of the spectrum are more susceptible to invasion by mesic weeds from adjoining Rainforests and shrubby Wet Sclerophyll Forests. There is scope to use fire to help reduce biomass of dense infestations, to deplete weed seeds in the soil seed bank and to facilitate competition by native species.

Fire and weeds in Dry Sclerophyll Forests

Dry Sclerophyll Forests shrub/grass subformation

Shrub/grass Dry Sclerophyll Forests are characterised by scleromorphic trees (typically eucalypts), with mixed semi-scleromorphic shrub and tussock grass understoreys. They are found on moderately fertile soils in moderate rainfall areas of the coast, tablelands and western slopes. The main plant families include Myrtaceae, Poaceae, Asteraceae, Epacridaeceae, Dilleniaceae and Fabaceae (Keith 2002).

Fire as a restoration tool

Kenny *et al.* (2003) recommend a fire interval of between 5 and 50 years, with occasional intervals greater than 25 years being desirable. The lower threshold reflects the dominance of grassy species in the understorey but moderate site fertility and rainfall mean that productivity is also moderate. These communities tend to be susceptible to invasion by both perennial exotic grasses as well as shrubs such as blackberry. Lantana is also a problem on the coast and escarpment in this subformation.

There is scope to use fire to help reduce biomass of dense infestations, deplete weed seeds in the soil seed bank and to facilitate competition by native species. In fragmented vegetation or areas subject to heavy grazing pressures, fire should be used with caution. In highly modified areas without a good source of species close by, hand sowing with native grasses and shrubs is advisable.

Dry Sclerophyll Forest shrubby subformation

Shrubby Dry Sclerophyll Forests are low forests of scleromorphic trees (typically eucalypts), with understoreys of scleromorphic shrubs and sparse groundcover. They are found on infertile sandy or shallow soils in regions receiving high to moderate rainfall on the coast, tablelands and low on the western slopes. The main plant families include Myrtaceae, Proteaceae, Epacridaceae, Fabaceae and Cyperaceae (Keith 2002).

<u>Fire as a restoration tool</u>

Kenny *et al.* (2003) recommend a fire interval of between 7 and 30 years, with occasional intervals greater than 25 years being desirable. Although these can be in areas subject to frequent wildfires they also have high numbers of obligate seeders which reach maturity slowly.

The post-fire environment can offer good opportunities to gain access to sites previously inaccessible due to heavy weed infestations or dense native shrub and vine layers. This is particularly the case with lantana in disturbed, logged and previously cleared sites. Gaining access post-fire can enable a targeted follow-up control of regenerating weeds, potentially resulting in a significant reduction in weed density and diversity and leading to improvements in the condition of these ecosystems. Practitioners using fire as a restoration tool in shrubby Dry Sclerophyll Forest should be cognisant of the maturation time of local obligate seeders if intending to use frequent fire.

Fire and weeds in Grasslands and Grassy Woodlands

Grassy Woodlands

Grassy Woodlands are open woodlands of scleromorphic trees (eucalypts, acacias, casuarinas), with open understoreys of xeromorphic shrubs, grasses and forbs, including many ephemeral species. They are found in areas with fine textured soils of moderate to high fertility and moderate rainfall from the tablelands to the western slopes with outliers in the drier coastal river valleys. The main plant families include Myrtaceae, Fabaceae, Myoporaceae, Asteraceae, Poaceae and Acanthaceae (Keith 2002).

Fire as a restoration tool

Kenny *et al.* (2003) recommend a fire interval of between 5 and 40 years, with minimum intervals of 10 years in the Southern Tablelands area. Occasional intervals greater than 15 years may be desirable.

Native Grasslands

Native Grasslands are closed tussock grasslands with a variable compliment of forbs. They are found on fertile soils of the tablelands and western floodplains. The main plant families include Poaceae, Asteraceae, Fabaceae, Geraniaceae and Chenopodiaceae (Keith 2002).

Fire as a restoration tool

Kenny *et al.* (2003) recommend a fire interval of between 2 and 10 years, with occasional intervals greater than 7 years in coastal areas.

Grasslands and Grassy Woodlands occupy some of the most productive and fertile landscapes. As a consequence they are amongst the mostly heavily cleared and fragmented ecosystems and often have a high proportion of weeds in the understorey.

Some success has been achieved through the use of fire regimes that advantage native grasses, herbs and forbs and disadvantage exotic varieties (G. Johnson, 2016 pers. comm; T. Dexter, 2016 pers. comm.). This is particularly the case where fire kills weeds and depletes or exhausts the availability of their seeds. Use of these fire regimes can contribute to a much greater dominance of native species post-fire. This ultimately depends on the mix of species present and other conditions, such as nutrient levels. Many native species in Grasslands and the understorey of Grassy Woodlands are advantaged by relatively frequent fire. On the other hand, relatively frequent fire kills, reduces seed numbers or diminishes the viability of many exotic grasses, herbs and other weeds.

Many exotic perennial grasses and some shrubs also respond well to fire, so an integrated approach that incorporates chemical or mechanical treatment may be required (Sanders *et al.* 2016). Careful consideration of the specific timing, sequencing and seasonality of these mixed fire and herbicide or mechanical control strategies is needed to ensure the best restoration outcomes. Sowing the post-fire ground, with native grasses and shrubs, following fire will assist with natural regeneration especially where exotic species have been dominant over native species for long periods as good sources of native plant seeds may not be present.

Fire and weeds in Forested Wetlands

Forested Wetlands are comprised of scleromorphic trees (eucalypts, paperbarks, casuarinas) with a sparse shrub strata and continuous groundcover of hydrophilous graminoids and forbs. They are found in flood-prone plains and riparian zones principally along the coast and inland rivers. The main plant families include Myrtaceae, Cyperaceae, Ranunculaceae, Blechnaceae and Poaceae (Keith 2002).

Fire as a restoration tool

Kenny *et al.* (2003) recommend a fire interval of between 7 and 35 years, with some intervals greater than 20 years being desirable for Swamp Sclerophyll Forests which include Forested Wetlands. They are characterised by the dominance of a number of trees including paperbarks (*Melaleuca* spp.), swamp mahogany (*E. robusta*) and swamp oak (*Casuarina* glauca), sometimes occurring as a monoculture. Forested Wetlands further inland are dominated by river oak (*C. cunninghamiana*) and, across the Murray–Darling Basin, by river red gum (*E. camaldulensis*). Here they occupy the main riverine channels and floodplains.

Fire regimes can vary greatly within Forested Wetlands due to their existence across a large climatic gradient from the wet coastal floodplain to the Paroo River at the western edge of the Murray–Darling Basin. Managing weeds with fire within forested wetland habitats can be challenging because of the wide variation in water availability and inundation levels, the presence of peat in many wetland complexes and the sensitivity to fire of many riparian and wetland plants and animals (including numerous threatened species).

In order to avoid damaging peat fires, if using fire as a restoration tool it should only be used when soil moisture is present. This negates the effectiveness of using fire to kill mature weed species or deplete weed seeds in the soil seed bank. Other methods of weed control will make a site more resilience to weed invasion in the event of a wildfire. Conducting follow-up control after wildfire is probably a better strategy in high rainfall areas.

Fire and weeds in Freshwater Wetlands

Freshwater Wetlands are characterised by swamp forests, wet shrublands or sedgelands, usually with a dense groundcover of graminoids. They are found throughout New South Wales on peaty soils with impeded drainage. The main plant families include Cyperaceae, Restionaceae, Juncaceae, Haloragaceae, Ranunculaceae and Myrtaceae (Keith 2002).

Fire as a restoration tool

Kenny *et al.* (2003) recommend a fire interval of between 6 and 35 years, with occasional intervals greater than 30 years being desirable. Similar constraints as detailed for Forested Wetlands apply in using prescribed fire as a part of a control strategy other than follow up control work after wildfires.

Fire and weeds in Heathlands

Heathlands are dense to open shrublands of small-leaved scleromorphic shrubs and sedges. They are found in high rainfall regions of the coast and tablelands on infertile soils, often in exposed topographic positions. The main plant families include Proteaceae, Fabaceae, Epacridaceae, Myrtaceae, Casuarinaceae and Cyperaceae (Keith 2002).

Fire as a restoration tool

Kenny *et al.* (2003) recommend a fire interval of between 7 and 30 years, with occasional intervals greater 20 years being desirable. Heathlands generally occupy sites with the lowest fertility and the poorest or shallowest soils and so may not have been subject to clearing for agriculture. On the coast, mining for mineral sands has modified large areas. On the tablelands, heathlands have been subject to mining for gold or base metals. These areas can be subject to weed invasion, particularly coastal areas where bitou bush was used for dune stabilisation, and from garden escapees.

In invaded coastal heath areas there is scope to use prescribed fire in an integrated way to help reduce the biomass of dense weed infestations, deplete seeds in the soil seed bank and to facilitate competition by native species.

8. Weed reviews

Methodology

This review undertook searches of all readily available sources of peer reviewed literature, 'grey' literature, unpublished reports and web resources and relevant expert knowledge was sought in order to access materials relevant to the interactions of fire and weeds within the native vegetation of New South Wales.

Peer reviewed journal articles as well as publications, materials and websites of government agencies, academic institutions, industry bodies and non-government organisations were the primary resources drawn on in the preparation of this review.

There is a general lack of specific knowledge of the interaction of fire with most of the weeds that are degrading the native vegetation of New South Wales. To our knowledge there have been no other reviews undertaken into the interactions of fire and weeds in the native vegetation of New South Wales at the statewide scale. There has been extensive interest in this review from across government and community and this interest is growing and ongoing.

Each of the weed species reviews includes the following information:

Status in particular if the weed is:

- a Weed of National Significance (WONS) listed in the 11 regional plans under the *NSW Biosecurity Act, 2015.*
- part of a key threatening process under the Biodiversity Conservation Act, 2016 which replaced the NSW Threatened Species Conservation Act (TSC Act, 1995)

Additionally:

- ecological attributes and preferred habitat
- interaction with fire
- key findings and management options

I. African lovegrass (*Eragrostis curvula*)

Existence in New South Wales

African lovegrass is a highly invasive perennial tussock forming grass which can reach around 1.5 metres in height. Its low palatability for grazing means that it becomes denser over time and forms thick dense swards which can exclude all other plant species. It is now found throughout New South Wales in a wide range of habitats, but particularly on the tablelands and South Coast.

Status

- *NSW Biosecurity Act 2015* - Listed as a Regional Priority Weed (Asset Protection) for the South East Region and a weed of Regional or Community Concern (Asset Protection) in the Central Tablelands, Greater Sydney, Hunter, Northern Tablelands and the Riverina in the various Regional Strategic Weed Management Plans. https://www.lls.nsw.gov.au/biosecurity/weed-control
- Key threatening process: *Invasion of native plant communities by exotic perennial grasses* (NSW SC 2003). Activities which facilitate or exacerbate the threat of invasion by African lovegrass to threatened species, populations and ecological communities, and their habitats must be considered when an activity requires assessment under Part 5A of the NSW *Environmental Planning and Assessment Act 1979* (EP&A Act).

Ecological attributes and preferred habitat

African lovegrass was originally introduced from South Africa prior to 1900. There were many additional introductions up until the late 1960s. It was planted as pasture grass and to stabilise soils in high erosion areas such as roadsides. It is spread by the movement of soil, machinery, slashing and in animal faeces (Firn 2009). It can survive and compete with plants in dry sandy soils of low fertility under a wide range of conditions. It grows in highly acidic conditions such as found in mine tailings where there is little competition. It can survive in areas of low rainfall and responds to grazing and increased nutrient inputs. It has the ability to keep growing during periods of drought until all moisture is gone, unlike other plants which appear to shut down when low moisture levels are reached (Firn 2009).

Growth slows in autumn and winter and it is susceptible to frost but plants do not die and will regrow again in spring as temperatures increase. It sets seed in summer and can produce large amounts of seeds (up to 600 kg/ha) with high viability (Johnston & Cregan

1979). Seeds are less than 5 mm in size and usually germinate in spring when soil temperatures are above 10°C. In warmer areas, seeds can germinate in any season if sufficient moisture is present. Seeds can be viable for up to 17 years in the soil (NSW DPI 2014a). Plants can spread vegetatively from tillers at the base after slashing or burning (Firn 2009).

Interaction with fire

A case study presented early results (Sanders *et al.* 2016) of a long-term trial at Cattai and Scheyville national parks on the Cumberland Plain, west of Sydney. It found that using a combination of fire, and spraying with the partially selective grass herbicide, Flupropanate, was effective in reducing the cover of African lovegrass. The most effective treatment was using fire with follow-up herbicide a year later then burning again the following year. The sites were colonised by a combination of native species as well as exotic annual weeds, but not African lovegrass.

It is suggested that the response of the other weed species was as a result of past heavy disturbance at the site and that better results would be expected at less disturbed sites where more native species are present (Sanders *et al.* 2016).

In the Bega Valley on the South Coast and the Monaro in the Southern Tablelands, the use of fire and Flupropanate is not recommended due to the potential impacts on non-target native grass species such as weeping grass (*Microlaena stipoides*) and the likelihood of areas being left as bare ground for long periods. This leaves the soil vulnerable to erosion and provides conditions most favoured by African lovegrass (J. Dorrough, 2016 pers. comm.). Instead a different methodology has been found to be cost effective in controlling African lovegrass on the South Coast in some areas. Grazing is used to reduce the height of more palatable grasses and then Glyphosate is applied with a roller wiper to the faster growing African lovegrass as it resprouts in spring (FSCLA 2016). On the Monaro, this approach is not recommended due to the rate of growth of target and non-target grasses being too similar and the rough terrain restricting the use of machinery (L. Pope, 2017 pers. comm.).

Weed management authorities through websites, factsheets and manuals provide some information about using fire to control African lovegrass. The NSW Department of Primary Industries website (DPI 2014a) advises that dead material in pastures can be removed by burning in late winter with follow up spot spraying with herbicide or mechanical removal. Most areas of remnant native pasture or grassy woodlands are likely to be on rough terrain which restricts the use of machinery. A Victorian best practice manual (Williams 2012) states that fire can be important in removing biomass for ease of cultivation and to assist in follow-up chemical control. It is also an effective tool in stimulating mass seed germination for chemical follow-up to reduce the soil seed bank. Fire should only be used in combination with a non-residual herbicide, as fire will reduce the effectiveness if a residual herbicide is being used to target seeds and roots. Burning may reduce the bulk of materials and allow other plant species to germinate, but may also result in periods of dry soils and loss of organic materials (NSW Industry and Investment 2010).

Areas dominated by African lovegrass can have overall fuel hazard loads in excess of 15 t/ha (FSCLA 2016). This creates an increased fire risk due to the intensity and rate of spread during wildfires (RFS 2012). Hazard reduction burns to reduce this risk are undertaken in winter when the grass is dried out and cured. This reduces the risk before warmer and windier conditions arrive in spring (Smythe 2016; RFS 2012).

Key findings and management options

- In higher rainfall areas, burning African lovegrass to reduce the sward density and stimulate native plant species and allow more targeted herbicide control can be effective (DPI 2014a).
- In areas of low to moderate rainfall where the soil may be exposed for long periods, the use of fire is less effective as replacement grass species take longer to establish. An integrated approach may include sowing of native pasture species to help restoration (DPI 2014a).

II. African olive (*Olea europaea* subsp. *cuspidata*)

Existence in New South Wales

African olive is a fast-growing, long-lived tree which can reach 15 m in height. It is highly invasive in a range of habitats from drier woodlands, riverine environments to coastal headlands and dune systems (Cuneo & Leishman 2006).

Status

- *NSW Biosecurity Act 2015* - Listed as a Regional Priority Weed for the Central Tablelands (Containment), Greater Sydney (Containment), Hunter (Asset Protection) and North West (Asset Protection) Regions. It is also a weed of Regional or Community Concern on the North Coast (Watch). https://www.lls.nsw.gov.au/biosecurity/weed-control
- Key threatening process: *Invasion of native plant communities by African olive* Olea europaea *L. subsp.* cuspidata (NSW SC 2010). Activities which facilitate or exacerbate the threat of invasion by African olive to threatened species, populations and ecological communities, and their habitats must be considered when an activity requires assessment under Part 5A of the EP&A Act.

Ecological attributes and preferred habitat

African olive is found naturally on volcanic soils in the Rift Valley and throughout eastern Africa. It also occurs sporadically across Asia to the arid areas of western China (Cuneo & Leishman 2006). Introduced to Australia as an ornamental and hedging plant, and unsuccessfully as root stock for commercial olive orchards, its fruits are not edible and it has no commercial value. The first plantings occurred at John MacArthur's farm near Camden, west Sydney, in 1820. Recent genetic analysis has shown that later introductions are the source of other populations of African olives in New South Wales (Besnard *et al.* 2014). Dense thickets were observed in western Sydney in the 1970s. In the 1980s its rate of spread increased dramatically and it is now a problem weed in the Hunter Valley and on the Illawarra coast. It is often found in disturbed areas and along the banks of drains and natural watercourses (Cuneo & Leishman 2006).

African olive leaves contain a high resin content which enables it to retain moisture during long dry periods. This enables it to compete with other plant species on sites with a western or northern aspect or that experience irregular rainfall. It can be survive in any area where the mean annual rainfall is over 800 mm.

It flowers in spring and the fruit ripens over winter. Mature trees produce fruit after 5–10 years on a 2–3 year cycle. Mature trees can produce over 25,000 fruits in a season. Fresh seeds have been recorded with viabilities of up to 88% but this appears to decline rapidly with most seed not germinating after nine months and few after two years in the soil (Cuneo, Offord & Leishman 2010) and (Cuneo and Leishman 2015). The fruit is spread by a range of frugivorous birds. It is thought that the partial removal of flesh by birds is important for dispersal and germination. Mammals such as the introduced European red fox (*Vulpes vulpes*) can also consume large quantities, but their importance as a vector is not known (Cuneo & Leishman 2006).

Interaction with fire

Fire will not kill mature African olive trees (Spennemann 1998). Large mature trees have been known to resprout 18 months after fire (Cuneo & Leishman 2006).

Von Richter, Little and Benson (2005) examined the mortality of small African olive trees after low intensity prescribed fire in Cumberland Plain Woodland vegetation. At this site, large mature individuals had been poisoned over several years but seedlings continued to germinate from the soil seed bank. A low intensity burn was conducted and the mortality of young trees was examined 12 months later. The study found a significant proportion of trees with stems less than 20 mm were killed by fire. This represented most plants less than 5 years old and some as old as 10 years. Water stress during periods of drought may also have contributed to the death of some trees. The study concluded that fire intervals of around ten years would be appropriate in this location to prevent the recruitment of small African olive trees (von Richter, Little & Benson 2005) as part of bushland restoration activities.

In another study in Cumberland Plain Woodland vegetation Cuneo and Leishman (2015) used fire to stimulate the germination of some native plant species in the soil seed bank after 15 years of the site being dominated by African olive.

Aires (2014) examined the potential for the presence of African olive to contribute to fuel loads in Cumberland Plain Woodland vegetation. He found that 'Overall, there was an increase in fine fuel loads, vertical distribution, fuel hazard score and flammability in areas densely invaded with African olive compared to more recently invaded areas and nearby pristine (non-invaded) woodland'. It is likely that moisture levels in dense thickets of African olive would restrict the spread of fire in most conditions, but in extremely dry conditions it could increase the intensity and duration of wild fires with the associated potential for increased damage to woodland ecosystems or nearby assets (Aires 2014).

Key findings and management options

- Fire is not effective as a control method for mature trees but young trees and seedlings are susceptible.
- Fire can be useful to stimulate the germination of seeds in the soil seed bank and the emerging seedlings can then be killed by a low intensity burn or other control methods**.**

III. Bitou bush (*Chrysanthemoides monilifera* subsp. *rotundata*)

Existence in New South Wales

Bitou bush (also called Bitou) has a near continuous distribution along the NSW coastal strip from Shoalhaven City Council to the Tweed Shire Council Local Government Areas (Australian Weeds Committee, 2012).

Status

- *Biosecurity* Act 2017 - Bitou bush is subject to a Biosecurity Zone for Containment under the New South Wales Biosecurity Regulation 2017 – “A biosecurity zone, to be known as the bitou bush biosecurity zone, is established for all land within the State except land within 10 kilometres of the mean high-water mark of the Pacific Ocean between Cape Byron in the north and Point Perpendicular in the South.” Listed as a Regional Priority Weed in Murray (Eradication) and Riverina (Eradication). It is also a weed of Regional or Community Concern in the Hunter. https://www.lls.nsw.gov.au/biosecurity/weed-control
- Weed of National Significance
- Key threatening process: *Invasion of native plant communities by bitou bush and boneseed* (NSW SC 1999). Activities which facilitate or exacerbate the threat of invasion by bitou bush to threatened species, populations and ecological communities, and their habitats must be considered when an activity requires assessment under Part 5A of the EP&A Act.
- A threat abatement plan was prepared for the invasion of native plant communities by *Chrysanthemoides monilifera* (bitou bush and boneseed) by the NSW Department of Environment and Conservation (DEC 2006).

Ecological attributes and preferred habitat

Bitou bush is distributed along most of the NSW coastline. It spread rapidly from both accidental and intentional introductions for coastal stabilisation and following sand mining. It came to dominate a large proportion of the coastal dune systems of the State, particularly on the North Coast (DEC 2006).

An aerial survey in 2002 (Thomas & Leys 2002) found bitou bush present along 900 km of the NSW coast (approximately 80% of the entire coastline). At the time this was an increase of approximately 240 km (or 36%) in the space of about 20 years (since the previous survey by Love in 1984). Thomas and Leys (2002) found bitou bush up to 10 km inland, noted that it dominated along 400 km of the coastline surveyed. They mapped and quantified bitou bush infestations along the coastline and estimated that 35,800 ha of public and private land were infested, including:

- 6700 ha heavily infested (with bitou bush dominant)
- 9000 ha medium infested (with bitou bush present but not dominant)
- 20,100 ha lightly infested with scattered plants.

In a follow-up survey in 2008, Hamilton *et al.* (2012) mapped significant reductions in the density and extent of both medium and heavy infestation density classes with reductions ranging from 87.5 – 96.7%. Bitou bush was recorded as absent from 4620 ha previously occupied by bitou bush in 2001. A slightly different methodology was employed for this study that included the addition of a “sparse “density class. This resulted in an increase in total area infested by bitou bush from 35,800 to 43,588 ha. When the same methodology was applied a decrease of 11% from 2001 to 2008 (36,408 ha to 32,274 ha) was recorded, including a 2% decrease in light density infestations Hamilton *et al* (2012).

In the Ministerial Foreword to the threat abatement plan for bitou bush (DEC 2006), Minister Debus wrote that bitou bush ‘…now poses the single greatest threat to NSW coastal ecosystems and coastal biodiversity, especially along the north coast. If it continues to expand unabated, within a decade there will be no area of the NSW coast unaffected. It forms dense infestations that smother sand dune, headland and hind dune vegetation communities including coastal grasslands, heathlands, woodlands, swamps/wetlands and forests’.

In recent years concerted control efforts, particularly those undertaken as part of the threat abatement plan have contributed to significant reductions in the distribution of bitou bush and limited the severity of many infestations and protected many populations of priority threatened species and patches of threatened ecological communities (Hamilton *et al.* 2010).

Bitou bush is a native of South Africa (Royal Botanic Gardens Trust 2016). The first herbarium specimen from New South Wales is dated 1908 and was collected from the Stockton area near Newcastle in New South Wales. It is assumed that it originated from ballast carried from South Africa (Gray 1976; Cooney, Gibbs & Golinski 1982). Boneseed (*C. monilifera* subsp. *monilifera*), the other subspecies of *Chrysanthemoides*, occurs on the

South Coast. Boneseed is a major problem in the temperate coastal habitats of Victoria, South Australia and Western Australia. Boneseed is regarded as a WONS in the same designation as bitou bush, as is also the case with the relevant key threatening process.

Bitou bush was deliberately planted by the NSW Soil Conservation Service to stabilise sand dunes along the NSW coast between 1946 and 1968 (DEC 2006). Bitou bush was also intentionally planted along the northern NSW coast to stabilise and revegetate coastal sand dunes following strip mining for the mineral sands rutile and zircon (Barr 1965). Bitou bush was recommended as one of several potential secondary stabilisers, including coastal tea tree and horsetail she-oak, to be planted following mining operations (Barr 1965).

Bitou bush is a fast growing perennial shrub up to 3 or 4 m in height and 6 m wide (Harden 1990). In sheltered sites stems may sprawl through supporting vegetation and reach lengths of up to 10 m. Classic yellow 'daisy form' inflorescences with 11–13 floral bracts are followed by black shiny fleshy fruit up to 10 mm in length containing a single seed up to 7 mm in length with a hard, woody endocarp. Up to 13 fruits are produced per inflorescence (Vranjjic 2000).

Bitou bush generally starts flowering 2–3 years after germination, but on the North Coast seedlings have been recorded flowering in their first year (DEC 2006). Flowering generally occurs between April and July, although Gosper (2004a) found a flowering peak between March and May. Peak fruiting is between June and September (Vranjic 2000), but once again Gosper (2004a), in a study at Illawarra on the south coast, found a slight variation and determined a peak of fruiting in May and June.

Seed production from a single mature bitou bush plant can be in the order or tens of thousands and seeds may be dormant and viable for up to 10 years (DEC 2006). Soil seed banks below mature thickets of bitou bush may reach 2000–5000 seeds/m^2 (Vranjic 2000).

Seed dispersal occurs with either the fruit falling from the parent plant (gravity dispersal) or via native frugivorous birds such as pied currawongs, Lewin's honeyeaters (*Meliphaga lewinii)* and silvereyes (*Zosterops lateralis*) (Dodkin & Gilmore 1984). At least 18 species of bird have been recorded consuming bitou bush fruits, most of which are likely to disperse the seeds (Gosper 2004b). The European red fox frequently feeds on and disperses bitou bush (Meek 1998). Because bitou bush produces large quantities of fruit during early winter when native fruits are scarce, it significantly alters the availability of this resource when compared to native ecosystems (Gosper 2004a). Birds that rely on nectar and fruit are less abundant within habitats invaded by bitou bush (French & Zubovic 1997; Gosper 2004b), whilst the assemblages and abundance of canopy foraging bird species and insectivores are largely unchanged within bitou infested habitats (French & Zubovic 1997; Gosper 2004b).

Germination of bitou seed can happen throughout the year, generally following precipitation, but fire can also promote germination when seeds are exposed to temperatures of 60°C. Removal of the seed coat also promotes germination (Weiss, Adair & Edwards 1998).

Bitou bush impacts significantly on numerous native plants and animals. These impacts are generated by bitou bush replacing and out-competing native plant species, physically excluding native plants and animals (Ens & French 2008), and by degrading and shifting the structural and floristic composition of various threatened ecological communities. Coutts-Smith and Downey (2006) identified 46 threatened entities as being threatened by bitou bush invasion. The threat abatement plan subsequently identified 157 plant species, 3 endangered plant populations and 24 ecological communities as being priorities for bitou bush control as a result of comments received during public exhibition of the plan and additional species modelling added another 70 species (DEC 2006; Hamilton, Winkler & Downey 2008).

Habitats invaded by bitou bush have altered invertebrate assemblages with species requiring higher moisture levels, such as springtails, millipedes, amphipods and slaters occurring in greater abundance (French & Eardley 1997; Lindsay & French 2004a, b). Less abundant groups are ants, earwigs, spiders and wood roaches. These changes in invertebrate assemblage are likely to have caused observed increases in decomposition rates and turnover of biomass (Lindsay & French 2004a, b).

Interactions with fire

The conditions of a fire (including the seasonality, weather and climatic conditions) and intensity can strongly influence the response of bitou bush to fire (Downey 1999). For example, moist soils can limit the soil temperatures attained during a fire, which directly influences the level of seed mortality and heat-stimulated germination that occurs (Downey 1999). Seed germination occurs from depths of up to 8 cm (majority up to 5 cm) in the absence of soil disturbance (Vranjic 2000). Following fire or mechanical damage (e.g. cutting), bitou bush has the ability to regenerate from adventitious buds at the base of the plant or along the stems, and vegetative reproduction can occur when the prostrate stems are buried by soil or sand (Weiss, Adair & Edwards 1998). Fire is recognised as being a key process in facilitating the spread or control of bitou bush (Weiss 1983; CRC for Australian Weed Management 2003). A hot fire will kill mature bitou bush whereas cooler fires will lead to resprouting, if the plant burns at all (Thomas *et al.* 2006). Hot fires kill bitou bush seeds in the upper parts of the soil seed bank but stimulate germination of much of the remaining soil seedbank (Thomas *et al.* 2006). If post-fire control is undertaken before bitou bush recruits reach maturity then ongoing infestations can be significantly reduced (Weiss 1983). If post-fire control of bitou bush recruits is not undertaken, such as was the case in the 1994 fires in central Yuraygir National Park, then bitou bush can significantly increase in abundance and expand the infestation (Flower & Clarke 2002 cited in Thomas *et al.* 2006).

In recent years a substantial body of literature has been published that addresses the responses of bitou bush to fire and other management interventions within experimental, adaptive management and opportunistic assessment frameworks (e.g. French *et al.* 2008; Vranjic *et al.* 2012; Lindenmayer *et al.* 2015). This research has enabled a much better understanding of the combinations of management interventions that achieve the best control outcomes for bitou bush and for cost-effectively restoring ecosystems infested with bitou bush (Lindenmayer *et al.* 2015).

Vranjic *et al.* (2012) experimentally investigated the effects of integrating revegetation with invasive plant management methods to rehabilitate coastal dune and woodland vegetation invaded by bitou bush. They found that fire increased densities of some native species in the woodland, but decreased those of others in the dune. Further significant findings of Vranjic *et al.* (2012) were that manual removal in both habitats and addition of seed in the woodland were most effective in reducing bitou bush densities when applied post-fire, and that herbicide treatment on its own or in combination with other treatments did not significantly reduce bitou bush densities by the end of the experiments. They concluded that 'restoration of coastal ecosystems invaded by a major invasive plant species requires a whole-of-system approach involving revegetation in combination with known management methods to assist recovery of native species in the longer term'.

Lindenmayer *et al.* (2015) undertook a 7-year experimental investigation of the effectiveness of various approaches and techniques for managing bitou bush at Booderee National Park, Jervis Bay. This research assessed quantified conservation benefits relative to management costs of different treatment regimes. Lindenmayer *et al.* (2015) examined a range of treatments including various combinations of spraying and burning with follow-up applications of each.

Spraying followed by burning and subsequent respraying was proven to be the most effective combination of treatment for reducing cover and abundance of bitou bush, whilst other regimes such as fire followed by spraying or two fires in succession were found to be less effective or were found to exacerbate bitou bush invasion. The spray-fire-spray regime was the most cost-effective approach to controlling a highly invasive species and facilitating restoration of native plant species richness to levels characteristic of uninvaded sites (Lindenmayer *et al.* 2015).

Logically, this long-term experiment determined that avoiding partial treatments and treatment sequences that exacerbate bitou bush impacts is critical to long-term restoration outcomes and that taking advantage of unplanned events such as wildfires can assist in achieving management objectives and reducing costs (Lindenmayer *et al.* 2015).

The experimental findings of Lindenmayer *et al.* (2015) are largely consistent with the bitou bush control results achieved within Bundjalung National Park on the North Coast. In the park an adaptive management response to an unplanned wildfire event allowed for management responses that led to significant reductions in bitou bush and substantial regeneration of native vegetation (Thomas *et al.* 2006).

The insights gained from these long-term experiments and adaptive restoration programs clearly demonstrate that combinations of herbicide treatments with fire can act to deplete seed resources within the landscape and achieve lasting weed control and ecological restoration outcomes.

Key findings and management options

- Having the capacity to adaptively respond to unplanned wildfire events can, in many instances, also assist in achieving good restoration outcomes. This may include allocating additional funding and resources for follow-up herbicide or mechanical control post-fire of germination, or control of mature patches unaffected by fire.
- Bitou bush has been successfully managed across a substantial proportion of the NSW coastline, mostly through the aerial application of low concentration herbicide, whilst in some landscapes, sequences of treatment incorporating fire and herbicide applications have resulted in good restoration and management outcomes.

IV. Blackberry (*Rubus fruticosus*) species aggregate

Existence in New South Wales

An aggregate of at least nine species of blackberry are found in New South Wales. It is highly invasive in a wide range of agricultural and natural ecosystems. Blackberry is a semi-deciduous, scrambling shrub with tangled, prickly stems that form impenetrable thickets several metres high. Thickets not only exclude native plants but also form an impenetrable barrier which restricts the movement of wildlife. While providing some habitat and food for native species, it also provides refuge for pest species such as cats (*Felis catus*), foxes and rabbits (*Oryctolagus cuniculus)*. It can also alter fire regimes by replacing grass in grassy ecosystems which would have previously carried fire more frequently (NSW DPI 2014c).

Status

- *NSW Biosecurity Act 2015* - Listed as a State Priority Weed (Asset Protection) and a priority weed for the Central Tablelands (Asset Protection), Hunter (Asset Protection), North West (Containment), and Northern Tablelands (Asset Protection) regions. It is also of community concern in the Central West, Murray, North Coast, Riverina and the South East. https://www.lls.nsw.gov.au/biosecurity/weed-control
- Weed of National Significance (Thorp & Lynch 2000).
- Key threatening process: *Loss and degradation of native plant and animal habitat by invasion of escaped garden plants, including aquatic plants* (NSW SC 2011). Activities which facilitate or exacerbate the threat of invasion by blackberry to threatened species, populations and ecological communities, and their habitats must be considered when an activity requires assessment under part 5A of the EP&A Act.

Ecological attributes and preferred habitat

All of the invasive blackberry species present in New South Wales are thought to originate in Europe. Introduced in the 1830s for use in gardens and hedges, by 1894 it had become a severe weed of farmland in New South Wales and Victoria (NSW DPI 2009). Blackberry is mostly restricted to areas with temperate climates (i.e. warm summers, cool winters) and an annual rainfall of at least 700 mm, but can grow in lower rainfall areas when sufficient moisture is available (such as along the banks of watercourses) (NSW DPI 2009).

Plants can spread both vegetatively and by seed. Each berry can contain from 20 to 30 seeds. At the end of the fruiting period (December to April), there may be up to 13,000 seeds/m^2 under a blackberry bush. Seed germination occurs in spring (NSW DPI 2009). Birds and a wide range of native and pest animal species can spread seeds over long distances, but water and movement of soil as part of agricultural and road building activities are also vectors. It spreads very quickly once established in a new area.

Blackberry seedlings are not vigorous in their first year but after a woody crown of around 20 cm is formed they become firmly established and growth increases (Ainsworth & Mahr 2006). They produce spreading stems known as primocanes. If a primocane touches the ground it may sprout roots and become an independent plant. Primocanes in turn sprout floricanes which produce flowers and fruit but die off the following autumn and winter. A new plant can also grow from root suckers or from root fragments if broken off and moved to a new site (NSW DPI 2009).

Interaction with fire

Burning will not kill blackberry but it can be used to make infestations more accessible for follow-up treatment (NSW DPI 2014c). The *Blackberry control manual* (NSW DPI 2009) states that the 'use of fire to control blackberry is generally ineffective: even though stems are destroyed, the woody crown and root system are only slightly affected'. It advises caution as burning may increase recruitment of seedlings and facilitate vegetative spread due to the reduced competition in the months after fire. It suggests that fire can be used to remove dead canes and material at least six months after treatment with herbicide. Burning areas to allow easier and more targeted application of herbicide may be appropriate in some areas. Wildfires may also open up areas and provide access to dense infestations (NSW DPI 2009).

Burning generally kills the seasonal canes but the root crown usually survives and regrowth can be quite vigorous after fire. Ainsworth and Mahr (2004) examined the ability of blackberry to respond after the high intensity bushfires experienced in eastern Victoria in 2003 to ascertain if the crowns and root systems were killed or produced less vigorous regrowth after a hot fire. They looked at a series of plots where different fire intensities were experienced and recorded the survival and regrowth of blackberry 12 months after the fire. They found that although a significant number of crowns were killed across a range of fire intensities, there were sufficient live root materials remaining which produced regrowth across all plots. Although the regrowth from root suckers was slow relative to normal growth rates, it was sufficient that in the post fire environment of high light and nutrients dense thickets would quickly re-establish (Ainsworth & Mahr 2004).

The site was re-examined in 2005, 2 years after the fire (Ainsworth & Mahr 2006). Blackberry was found to have re-established at many sites, but not all of them. At some sites, native shrubs and eucalypt tree seedlings were out-competing the blackberry. There was no correlation with fire intensity. At one site the presence of shallow stony soil may have helped native species to compete with the blackberry. The authors concluded that rainfall may also have been a factor but that if there is a good source of native plant seeds then native shrubs and trees can compete with blackberry. They recommend trials of heavy seeding with local native shrubs after bushfires in areas where the native soil seed bank may be depleted (Ainsworth & Mahr 2006).

Davies (1998) recommended research into: how native species compete with blackberry, the condition of native seed banks under blackberry thickets, and the use of fire to stimulate native seed germination under and around thickets.

Key findings and management options

- Fire will not kill Blackberry roots crowns but can be useful in dense infestations to remove dead canes and material after initial herbicide treatment
- In some areas fire is useful to reduce the density of large thickets of blackberry to allow access for other control methods (NSW DPI 2009).

V. Boneseed (*Chrysanthemoides monilifera* subsp. *monilifera*)

Existence in New South Wales

Boneseed is a highly invasive perennial shrub reaching about 3–4 m in height (Lane 1976). It is one of two subspecies present in Australia the other being bitou bush (see separate review).

Status

- *Biosecurity Act 2017* - Listed as a State Priority Weed (Eradicate) - Control Order 2017 for the whole state. https://www.lls.nsw.gov.au/biosecurity/weed-control
- Weed of National Significance (Thorp & Lynch 2000).
- Key threatening process: *Invasion of native plant communities by bitou bush and boneseed* (NSW SC 1999). Activities which facilitate or exacerbate the threat of invasion by boneseed to threatened species, populations and ecological communities, and their habitats must be considered when an activity requires assessment under Part 5A of the EP&A Act.

Ecological attributes and preferred habitat

Boneseed originates from south-west South Africa. It is not known exactly when and where boneseed was first introduced to Australia, but it was recorded in a Sydney garden in 1852 (Gray 1976). The infestations of boneseed in New South Wales are thought to be all garden escapees as only bitou bush has been recorded as being used to stabilise coastal sand dunes after mining in New South Wales (Weiss *et al.* 2008). It is grazed by stock so it is not a problem weed of agricultural grazing land (Groves 1990).

It is naturalised in coastal districts from the Hunter River to Moruya on the South Coast. There are also populations in the Blue Mountains, western New South Wales around Broken Hill and at Dareton on the Murray River. Unlike bitou bush, it has also invaded inland areas where sandy soils are present (Brougham *et al.* 2006). Boneseed prefers areas which receive most of their annual rainfall in winter. It is found in a wide range of vegetation communities including coastal dunes, estuarine areas, heath, mallee, woodland, and dry and wet sclerophyll forests (Brougham, Cherry & Downey 2006).

Boneseed does not spread vegetatively and relies on seed dispersal. It flowers annually between autumn and spring and can produce up to 50,000 seeds/m^2 each year. The fruits are spread by birds and mammals and can germinate anytime of the year. Young plants usually take 18 months to mature and flower but this can be longer or shorter depending on site conditions. The seeds develop a hard coat which enables them to roll down slopes or float in sea water to germinate at new locations (Brougham, Cherry & Downey 2006). They can also survive cool fires. After being exposed to the elements, three cracks form and the seeds are ready to germinate when sufficient moisture is present (Lane & Shaw 1978). Seeds are viable in the soil for at least 8.5 years (Briden & McAlpine 2012).

Interaction with fire

Lane and Shaw (1978) found that exposure to temperatures of 100°C for 30 seconds was sufficient to stimulate the germination of seeds in the laboratory and in field trials. However, seeds which had not been buried for at least a season had not weathered sufficiently to germinate. At temperatures of 150°C seeds were killed after an exposure time of 8 minutes. At 250°C seeds were killed after only 2 minutes. They recommend fire as a control method but cautioned that damage may occur to native species. Surviving native vegetation would, however, have a better chance of survival when boneseed is removed or controlled.

Noble and Weiss (1989) looked at the movement of seeds in the soil in the context of using biological control in the form of a seed predator. The study used seeds from bitou bush however the conclusion regarding the potential use of intensive fire to enhance the effects of biological control is also applicable to Boneseed. The feasibility of using fire in combination with a biological agent would be dependent on the impact of fire on each stage of its life cycle.

Groves (1990) stated that fire regimes could be used to help control boneseed by using a follow-up prescribed burn after a wildfire to kill seedlings before they flowered. This would reduce the seeds left in the soil seed bank. Potential impacts on native species would need to be assessed before implementation.

Melland and Preston (2008) recommend that fire can play an important role as part of an integrated approach to eradicating or controlling boneseed by: killing adult plants, providing access to areas of dense infestation, and depleting the soil seed bank by killing seeds and triggering mass germination for follow-up control. Unlike bitou bush, boneseed does not resprout after fire so if it is subject to total leaf scorch it can be killed by fire.

Briden and McAlpine (2012) compared the regeneration of boneseed seedlings in 78 plots after a wildfire in New Zealand burnt through an area where boneseed had been manually removed 8.5 years earlier. A total of 172 seedlings were present in the burnt plots and only 1 in the unburnt area. The finding that seeds were viable in the soil for at least 8.5 years is consistent with estimates of 10 years in Australia (Brougham *et al.* 2006).

The *Boneseed management manual* (Brougham *et al.* 2006) has a section on the use of fire as a part of control actions. The following is a summary of this information in regards to the interaction of boneseed and fire. A hot fire can kill many seeds stored in the soil, but will also stimulate mass germination of seedlings which will form a dense carpet on bare ground. If this mass germination is controlled within the first year, before plants reach maturity and fruit, the soil seedbank can be significantly depleted. If this does not occur any germination of native seedlings will be quickly out-competed (Brougham, Cherry & Downey 2006).

Boneseed does not burn well under normal growing conditions. Dense thickets over a metre in height restrict the growth of more flammable grasses and herbs which would help the spread of fire and increase its intensity enough to kill adult plants and deplete the seed soil bank. The importance of getting a hot fire is emphasised. This can be achieved by cutting large individuals and spreading them out 12 months before a prescribed burn. This will help provide dry fuel for the burn in the ground layer. This may increase the bush fire risk so safeguards should be put in place. Be ready for weed control measures to be implemented after wildfires to remove new seedlings via hand pulling or herbicide use. The importance of an even burn is also highlighted to ensure a good germination of seedlings occurs to deplete the soil seedbank (Brougham, Cherry & Downey 2006).

Key findings and management options

- Fire of at least moderate intensity can be used to kill mature and young plants. This allows access to dense areas for future control and kills seeds in the soil seed bank and triggers mass germination which can be killed by a follow-up burn or other control techniques (Brougham, Cherry & Downey 2006).
- Mechanical felling and drying of large mature trees may be required to achieve an effective burn.
- Follow up control of emerging seeds may be required for at least 10 years.

VI. Camphor laurel (*Cinnamomum camphora*)

Existence in New South Wales

Camphor laurel is limited to Rainforests, adjoining Wet and Dry Sclerophyll Forests (and occasionally Forested Wetlands and Heathlands). It is dominant across the cleared, high-fertility rainforest landscapes in the higher rainfall coastal river valleys of central and north-east New South Wales (e.g. the Big Scrub in the coastal parts of the Richmond Valley).

Status

- *Biosecurity Act 2017* - Listed as of Regional Concern or Community Concern in Greater Sydney, Hunter and North Coast (Asset Protection) Regional Plans. https://www.lls.nsw.gov.au/biosecurity/weed-control

Ecological attributes and preferred habitat

Camphor laurel occurs along the coastal north-eastern fringe of New South Wales, mostly in fertile river valleys along the North Coast and Mid North Coast in areas that experience high annual rainfall, generally in excess of 1000 mm. Camphor laurel grows on a wide range of soil types, but reaches best establishment and dominance on fertile floodplain alluvium and soils derived from basalt (Big Scrub Rainforest Landcare Group 2005; Stubbs 2012)

Camphor laurel establishes and dominates across landscapes where forests have been cleared or disturbed, usually for pasture or cultivation, and particularly on former dairy farms which originally supported rainforest vegetation (Lymburner, Handley & Handley 2006).

Within forested areas camphor laurel generally only recruits along forest margins, cleared river banks, tracks and roadways and after disturbances such as logging, herbicide application and fire (M. Graham 1996–2016, pers. obs.). In places such as the Bellinger, Orara and Richmond Valleys extensive groves of camphor laurel provide an abundant source of seed (M. Graham 1996–2016, pers. obs.).

Camphor laurel was introduced as early as the late 1820s into botanic gardens, such as the Royal Botanic Gardens in Sydney, and as a shade tree for street, school and other municipal purposes (Stubbs 2012). It was then distributed for planting across towns and cities in the subtropical zone of northern New South Wales and southern Queensland.

Camphor laurel is a tall (greater than 30 m) and long-lived rainforest tree that can potentially live for many centuries. Mature camphor laurel trees bear many thousands (potentially hundreds of thousands) of oily fleshy black fruit, ripening in autumn and winter when many of the closely related native laurels that occur in the rainforests of northern New South Wales lack fruit (Firth 1979). It is dispersed by several native species of frugivorous bird, often those that feed on the numerous native laurels and other native fruit-bearing rainforest species that occur within landscapes that support the camphor laurel infestations (Neilan *et al.* 2006).

Whilst dominating in higher fertility landscapes, camphor laurel will establish within less fertile and drier landscapes (e.g. the sandstone landscapes of the coastal Clarence Valley). In these landscapes, the slower growth rates mean trees will reach reproductive maturity much more slowly.

After germination, camphor laurel invests significant energy in root establishment, often forming a tap root structure. This structure enables small camphor laurel seedlings and sapling to survive fire by resprouting and suckering.

Interactions with fire

Mature camphor laurel trees have been documented suckering profusely after being individually burnt, but there are few documented instances of broadscale fire within landscapes dominated by camphor laurel.

Camphor laurel is a rainforest tree which occupies rainforest landscapes that are generally moist and not prone to fire. It plays an important role in facilitating rainforest establishment in a landscape. As such, fire is generally not a significant process in relation to its life history.

With recruitment of camphor laurel seedlings and establishment within Wet Sclerophyll Forests and other habitats in which fire plays a greater role in ecosystem function (e.g. Dry Sclerophyll Forests and Heathlands on the Far North Coast in landscapes where seed supply is abundant), the main opportunity to manage camphor laurel is to use fire to kill seedlings and saplings and prevent establishment and expansion of camphor laurel into these habitats. As camphor laurel has a hardy root system and a propensity to sucker prolifically, it is likely that the application of follow-up regeneration work will be required post-fire.

Key findings and management options

- Fire should only be used to prevent recruitment and establishment of camphor laurel within habitats in which fire is appropriate.

- The use of fire to control camphor laurel within rainforest habitats is generally not appropriate, although in the event of wildfire within rainforest habitats invaded by camphor laurel, it is recommended that post-fire control be undertaken using appropriate techniques such as foliar spraying and stem injection to prevent rapid establishment and dominance of the habitat by camphor laurel post-fire.

VII. Chilean needle grass (*Nassella neesiana*)

Existence in New South Wales

Chilean needle grass is a highly invasive perennial grass of natural grassy ecosystems and agricultural land. It is found from the Northern Tablelands and along the Great Dividing Range to Victoria (NSW DPI 2015). It is relatively unpalatable for grazing animals, especially after it sets seed, due to the presence of long sharp needle-like seeds. It is closely related to another significant weed, serrated tussock (*Nassella trichotoma*) (Vic. DPI 2007).

Status

- *NSW Biosecurity Act 2015* - Listed as a State-wide Priority Weed (Asset Protection) and a Priority weed for the Central Tablelands (Containment), Central West (Containment), Hunter (Prevention), Murray (Containment), North West (Containment), Northern Tablelands (Asset Protection), Riverina (Eradication) and South East (Asset Protection) regions. It is also a potential risk for the North Coast (Watch) region. https://www.lls.nsw.gov.au/biosecurity/weed-control
- Weed of National Significance (Thorp & Lynch 2000).
- Key threatening process: *Invasion of native plant communities by exotic perennial grasses* (NSW SC 2003). Activities which facilitate or exacerbate the threat of invasion by Chilean needle grass to threatened species, populations and ecological communities, and their habitats must be considered when an activity requires assessment under Part 5A of the EP&A Act.

Ecological attributes and preferred habitat

Chilean needle grass is originally from the Pampas Grasslands of South America. It may have evolved with predation from extinct megafauna (Faithfull 2012). It was first identified at Glen Innes in 1943, probably from an accidental introduction. Its rate of spread was slow until the 1970s (Benson & McDougall 2005).

It grows in dense clumps or tussocks of up to a 1.5 m in height which can live for more than 20 years and is spread mainly by adhering to livestock, clothing or machinery (Benson & McDougall 2005). It can compete readily in areas where annual rainfall is between 500 and 1000 mm. In New South Wales it has become established on the more fertile soils but in other areas it competes well in soils of low fertility (Vic. DPI 2007).

It grows in all seasons except summer. Across most of its range it flowers in December before setting seed in the panicle. In northern New South Wales it also flowers again in autumn. It also produces hidden seeds, known as cleistogenes, which form in nodes at the base of the leaf and main stems. These enable the plant to reproduce despite the upper panicle being removed by grazing, slashing or fire. Plants can produce seeds in the first year which are viable in the soil for at least 3 years and possibly for more than 12 years (Benson & McDougall 2005). It needs bare ground to establish and can tolerate drought and heavy grazing (Vic. DPI 2007).

Chilean needle grass can produce 22,000 seeds/m^2 annually, of which almost half become part of the soil seed bank (Whalley, Andrews & Gardener 1997). Chilean needle grass prefers open areas but it will grow in grassy woodlands. It can tolerate seasonal waterlogging (Faithfull 2009).

Interaction with fire

Faithfull (2009) examined the invasion of native grasslands by Chilean needle grass. He postulated that burning off grasslands dominated by the summer growing kangaroo grass (*Themeda triandra*) in autumn and winter rather than spring would leave the way open for the winter growing Chilean needle grass to capture more light, nutrients and space.

Trials in Victoria found that burning in early and late spring resulted in less than 10% of Chilean needle grass resprouting when growth began again in winter. This was a reduction of 75% when compared to unburnt sites. There was however an increase in the number of small and immature tussocks, probably as a result of cleistogene seeds in the burnt stubble germinating. A late spring burn ‘removed all viable seed from the site’ and reduced viable seed production by 50% (Faithfull 2009). Fire can destroy seeds in the upper part of the plant but the basal stem seeds usually survive and enter the seed soil bank (Benson & McDougall 2005).

Chilean needle grass does not readily invade native grasslands if dense swards of kangaroo grass area present. Sowing seeds of kangaroo grass and other grassland species, after using fire to reduce and destroy fallen Chilean needle grass seed, is recommended as a control method (Faithfull 2009). Bourdôt (2010) recommends the removal of tussocks by mechanical means and then spot spraying with glyphosate followed by burning and then a second glyphosate treatment to kill emerging seedlings.

The National Best Practice Management Manual for Chilean Needle Grass (Vic. DPI 2007) states that fire can be used as part of an integrated weed program to prevent seed set, burn off standing seed and stimulate the growth of seeds in the soil seed bank. Fire will stimulate the germination of native grassland species. Follow-up herbicide spraying of Chilean needle grass will assist native species to compete.

A case study was presented from Victoria where herbicide treatment of Chilean needle grass was combined with ecological burning and some supplementary sowing of gaps with native grasses in spring. The native species have been able to compete with the Chilean needle grass and with ongoing control it has been reduced each year (Vic. DPI 2007).

Key findings and management options

- At sites where summer growing native grasses are present, burning in spring can assist in competing with the winter growing Chilean needle grass and reduce the resprouting success of mature plants. Follow-up control of new seedlings will be required as seeds not killed by fire will later germinate if the ground remains or becomes bare. Sowing gaps with native grasses will assist recovery (Vic. DPI 2007).

VIII. Coolatai grass (*Hyparrhenia hirta*)

Existence in New South Wales

Coolatai grass is an exotic species with a relatively widespread distribution across the north of New South Wales, with the core distribution on the north-west slopes and plains.

Status

- *NSW Biosecurity Act 2015* - Listed as a State-wide Priority Weed (Asset Protection) and a priority weed for the South East (Containment) region. Weed of regional concern in the Hunter and North Coast regions. https://www.lls.nsw.gov.au/biosecurity/weed-control

Ecological attributes and preferred habitat

Coolatai grass occurs across a broad range of landscapes, soil and habitat types in northern New South Wales from sea level to 1500 m ASL and across a broad climatic range from high rainfall subtropical coastal valleys to semi-arid parts of the Murray–Darling Basin. Coolatai grass is a weed that has spread rapidly and colonised a large area, including intact native vegetation communities with a grassy understorey such as critically endangered Native Grasslands and Grassy Box-Gum Woodlands (McArdle, Nadolny & Sindel 2004).

In many parts of its core distribution on the north-west slopes and plains, Coolatai grass is the dominant groundcover across whole landscapes, covering travelling stock reserves, farmlands, roadsides and conservation reserves with a large biomass and near complete coverage of the ground layer. This situation is rapidly worsening as Coolatai grass continues to invade new sites.

Coolatai grass is a native of tropical and temperate zones between Africa, the Mediterranean region, the Middle East and India. It is likely to have originally been introduced from southern Africa to Queensland and northern New South Wales in the late 1800s. The distribution of the species remained relatively limited until the 1950s when widespread introduction by CSIRO and state government agricultural departments occurred, particularly across northern New South Wales and southern Queensland. Coolatai is a hardy perennial grass that survives heavy grazing well (CRC for Australian Weed Management 2007).

Capable of invading undisturbed native Grasslands and Grassy Woodlands, Coolatai grass is a serious modifier of these high conservation value grassy ecosystems (McArdle, Nadolny & Sindel 2004; Chejara *et al.* 2006). Across large parts of the north-west slopes and plains that have been heavily invaded, Coolatai Grass is suspected to have been responsible for causing major population declines in native granivorous birds, including finches and doves, as well as native rodents and a significant reduction in reptile diversity (Phil Spark 2002, pers. comm.). This is a result of the grass causing fundamental changes to the vegetation structure and a reduction in plant diversity.

Coolatai grass grows vigorously, forming an almost complete monoculture and replacing native grass and wildflower species. It tolerates drought, heavy grazing and many herbicides. It has invaded large areas of grassy woodlands and native pastures in north-west New South Wales and is spreading rapidly in other regions (CRC for Australian Weed Management 2007).

Storrie (2003) writes:

‘Coolatai grass poses a huge risk to the biodiversity of the fragmented areas of native ecosystems remaining across New South Wales as it easily invades relatively undisturbed ecosystems. The mechanisms of how this occurs are still not fully understood but Coolatai grass has a number of characteristics that allow it to invade a range of ecosystems:

- plants are long lived
- able to produce fertile seed from a single plant
- seed is mobile – wind, water, animals, vehicles
- seed will germinate over a wide range of temperatures
- seeds are able to germinate and establish at the soil surface in the presence of leaf litter
- established plants are tolerant of drought, fire and herbicides.’

In short, Coolatai grass is a major ongoing threat to the biodiversity of the north-west slopes and plains; a region with a high level of native vegetation clearance, fragmentation and degradation and poor long-term prospects for biodiversity conservation. Many nationally listed species and communities are threatened by Coolatai grass.

Monitoring of native fauna in areas dominated by Coolatai grass near Manilla, New South Wales, found that the abundance of ground-active invertebrates was reduced. Preliminary results suggest that the diversity and abundance of reptiles and frogs were also reduced

(Spark 2007, in CRC for Australian Weed Management 2007). It is also feared that Coolatai grass invasion will cause further declines of a number of ground-feeding vulnerable woodland birds, including, hooded robin (*Melanodryas cucullata cucullata)*, brown treecreeper (*Climacteris picumnus victoriae)*, turquoise parrot (*Neophema pulchella)*, speckled warbler (*Chthonicola sagittata)* and diamond firetail (*Stagonopleura guttata)* (Spark 2007, in CRC for Australian Weed Management 2007).

Interactions with fire

Coolatai grass produces substantial amounts of biomass that has the potential to dry quickly and significantly increase fuel loads. This fuel contributes to fire regimes that favour it over native species. Hotter and more frequent fires attributable to Coolatai grass infestations have caused changes in vegetation structure and diversity and lead to local extinctions of many native plant and animal species (McArdle, Nadolny & Sindel 2004; Chejara *et al.* 2006).

Coolatai grass rapidly regrows following fire and is positively influenced by fire. Post fire growth is promoted by fire, and seed production can increase in the seasons post-fire (McCormick, Lodge & McGufficke 2002).

Key findings and management options

- The use of fire alone as a management tool for Coolatai grass should be avoided.
- Carefully planned management strategies that integrate the use of fire with herbicide removal are the only instances in which the use of fire should ever be considered for managing Coolatai grass, but because fire promotes Coolatai grass even these strategies are not likely to be effective in managing the species.
- Mechanical, grazing and manual control strategies combined with herbicide application are recommended for best control outcomes.

IX. Lantana (*Lantana camara*)

Existence in New South Wales

Present across all coastal parts of New South Wales, but particularly widespread and abundant north of the Shoalhaven Valley. Absent from the higher parts of the tablelands, but has the potential to expand its range southward as well as inland and into higher elevation landscapes as a consequence of global warming (Goncalves *et al.* 2014*;* Taylor *et al*. 2012).

Status

- *NSW Biosecurity Act 2015* - Listed as a State-wide priority weed (Asset Protection). Priority weed for the South East (Containment) region. Weed of regional concern in Greater Sydney (Asset Protection) and the North Coast (Asset Protection) regions. https://www.lls.nsw.gov.au/biosecurity/weed-control
- Weed of National Significance (Thorp & Lynch 2000).
- Key threatening process: *Invasion, establishment and spread of Lantana (*Lantana camara) *L.* sens. lat*)* (NSW SC 2006)
- *A Plan to protect environmental assets from lantana* (Biosecurity Queensland 2010) constitutes a threat abatement plan for lantana.

Ecological attributes and preferred habitat

Lantana is regarded as a weed of international significance (Sharma, Raghubanshi & Singh 2005) because of its widespread distribution and abundance across temperate, subtropical and tropical climatic zones (Swarbrick 1986; QLD NRM&E 2004) and its myriad negative impacts on primary production and biodiversity (Sharma, Raghubanshi & Singh 2005).

Lantana has invaded more than 5% of the Australian continent, and is a Weed of National Significance that impacts greatly on biodiversity (Biosecurity Queensland 2010). Lantana currently infests more than 4 million hectares of land across Australia, mainly in areas east of the Great Dividing Range in New South Wales and Queensland. Its current range extends from the Bega Valley Shire in southern New South Wales to Cape Melville in north Queensland.

In New South Wales, lantana is widespread across coastal valleys and is continuing to invade new habitats within its current range and to increase its density within existing infestations (DPI 2015b).

Lantana is a native of the tropical and subtropical regions of Central and South America (Royal Botanic Gardens Trust 2016). Lantana was first introduced into Australia as an ornamental plant in 1841 and by the 1860s it was naturalised in the Sydney and Brisbane areas.

Lantana grows across a wide range of coastal and subcoastal landscapes, rapidly establishing and dominating in humid coastal conditions. Its greatest development and dominance is achieved in areas with an average annual rainfall exceeding 900 mm. Lantana is frost sensitive and stops growing when temperatures fall below 5°C; and because of this sensitivity it has an altitudinal range in New South Wales of approximately 1000 m ASL which may increase with global warming but this is also dependent on other factors such as moisture availability and seed dispersal (Taylor *et al*. 2012). Lantana has spread extensively along the east coast of Australia, with whole ecosystems and many species now threatened (Turner, Hamilton & Downey 2008; Turner & Downey 2010)

Lantana prefers moist soils in landscapes dominated by rainforests and tall shrubby Wet Sclerophyll Forests. Lantana can also recruit and establish within Dry Sclerophyll Forests (anecdotally, this trend seems to be increasing over time) and in such habitats can survive prolonged dry periods with physiological responses including dormancy and wilting.

Lantana reaches its greatest development in well-drained, fertile soils including rich organic soils, well-drained clay soils and volcanic soils. It tolerates poor soils and sand and will grow on stony hillsides where moisture is available. Lantana is tolerant of partial shading, persisting within relatively low light conditions, but will not establish within intact forests subjected to complete shading.

Lantana is a major weed along roadsides, riparian zones (river banks), fencelines, forestry areas, pastures and waste areas. It also invades open native woodlands and sub-tropical rainforest fringes and often can grow in steep inaccessible areas where it reproduces vegetatively via layering canes, making control difficult. Lantana readily invades disturbed or unmanaged areas or where native forests and woodlands have been thinned or cleared for grazing (DPI 2015b). Lantana is less common in undisturbed native vegetation communities. In optimal conditions lantana forms dense stands of greater than 5 m height that physically prevent the establishment of native seedlings (Swarbrick, Willson & Hannan-Jones 1998), or inhibit and reduce the germination, seedling growth and survival of native plants (Fensham, Fairfax & Cannell 1994; Gentle & Duggin 1997a, 1998; Sharma, Raghubanshi & Singh 2005).

There are at least 29 known varieties of lantana with a range of flower colours from pink to mauve and red. There are subtle differences in growth habit and rates of establishment between these varieties (DPI 2015b). All varieties bear dark shiny black fleshy fruit that are dispersed considerable distances by a diversity of native bird and mammal species. A mature lantana plant can produce up to 12,000 seeds/annum and dispersal distances of greater than 1 km are known (Swarbrick, Willson & Hannan-Jones 1998). Lantana plants can live for multiple decades and plants can rapidly resprout following defoliation by frost or fire (Swarbrick, Willson & Hannan-Jones 1998; QLD NRM&E 2004).

Bell miner associated dieback (BMAD) is a particularly insidious and recently rapidly spreading ecologically degrading process that is strongly associated with lantana infestations occurring within eucalypt forests and woodlands along the east coast (Silver, MJ and Carnegie AJ, 2017). BMAD is a form of ecological 'meltdown' in which super-abundant colonies of bell miner (*Manorina melanophrys*) establish within forests degraded by lantana and exert a degrading influence on eucalypt forest canopy health though 'farming' lerp from high density infestations of psyllid (Wardell-Johnson *et al* 2006). This process ultimately leads to the death of dominant eucalypts, a major loss of biodiversity and a situation in which lantana dominates in what were previously biodiverse native forests (BMAD Working Group 2004; Somerville, Somerville & Coyle 2011). In a limited subset of BMAD afflicted sites at higher elevations at which lantana is absent, dense "cloaks" of vines including native *Cissus* spp. provide the structural habitat component required for the establishment of bell miner colonies (BMAD Working Group 2004)(Silver, MJ and Carnegie AJ, 2017).

In a statewide analysis of the degree of threat posed by weeds to threatened species in New South Wales, lantana was found to pose a threat to the greatest number of listed threatened entities (i.e. 96) of any weed occurring in the State (Coutts-Smith & Downey 2006). The fundamental ecosystem transformation that lantana creates (see Richardson *et al.* 2000) creates impacts on a myriad of other non-listed species of flora and fauna and numerous native vegetation formations, subformations, classes, communities and associations.

Interactions with fire

The leaves and canes of lantana are susceptible to fire, but canes have the potential to resprout rapidly following minor scorching and in situations of less than complete combustion. The conditions of a fire including the seasonality, weather and climatic conditions and intensity can strongly influence the response of lantana to fire, and generally only fires of medium to high intensity exert any appreciable controlling influence on adult plants.

Intense spring and summer wildfires have been noted to result in a significant reduction in dense lantana cover in tall wet sclerophyll forests in the Bellinger and Orara valleys of northern New South Wales (M. Graham, pers. comm.), but the seasonal conditions following the fire determine the degree to which the extent and density of lantana is permanently reduced. In one notable instance of a hot spring fire in the Orara Valley, a lengthy post-fire period with little or no rainfall (about 4 months) resulted in a permanent loss of extensive lantana patches and replacement with a diversity of native species (M. Graham, pers. comm.). This particular case is very much the exception and not the rule when it comes to lantana and fire in northern New South Wales.

Gentle and Duggin (1997b) identified fire as a significant disturbance that facilitated the invasion of lantana into a site. Raizada and Raghubanshi (2010) found that smoke increased the success of lantana seed germination and lowered subsequent lantana seedling mortality. When subject to total leaf scorch Lantana recovers by shooting from basal dormant buds (Swarbrick, Willson & Hannan-Jones 1998), so follow-up control with manual, mechanical and herbicide techniques is necessary to ensure the death of large plants and seedlings which have germinated following wildfire or a prescribed burn (NSW DPI 2015b).

Under varying conditions lantana will either increase or decrease the fuel load along a rainforest margin (Swarbrick, Willson & Hannan-Jones 1998). In these areas fire is not appropriate, because lantana provides a fuel load that changes the intensity of fire and degrades the adjoining ecosystem (Day *et al.* 2003). As a means to reduce the invasion hazard of lantana, Duggin and Gentle (1998) suggested that fire should be completely removed from the ecotones between dry rainforest and open forest in northern New South Wales, except for low-intensity fires to manage fuel loads.

With lantana frequently recruiting into Dry Sclerophyll Forests and Grassy Woodlands (and anecdotally increasing its propensity to do so) that are adapted to a relatively high fire frequency, there are numerous examples of where regular burning has acted to prevent the establishment and dominance of lantana within these ecosystems. Many such instances are the use of fire to promote green pick for grazing within bushland areas at the fringes of grazing properties. Application of these types of fire regimes has some potential to limit the spread and reduce the severity of lantana invasions within a landscape.

Key findings and management options

- The many interactions of fire and lantana can often be a 'double-edged sword' and a process that will require cautious and thoughtful management to avoid exacerbating the degrading influence of this transformer weed. In landscapes supporting ecosystems adapted to a relatively high fire frequency there is good potential for the use of fire to prevent the establishment of lantana recruits and to limit its spread at the landscape scale.
- As dense infestations of lantana are generally moist, there are few opportunities for achieving the ignition required for managed fire. Generally when such infestations are dry enough to burn is when prevailing seasonal conditions are such that few are willing (or brave enough) or indeed legally permitted to use fire. At these times the fire intensity achievable has good potential as a tool to achieve a significant kill of lantana within these dense patches. In the event of wildfires in such conditions there can be excellent opportunities to adaptively respond to the situation and to follow-up with appropriate restoration techniques and achieve broader-scale and longer-lasting lantana control outcomes.
- In many landscape positions and ecological contexts in which lantana occurs the surrounding vegetation is rainforest, in these situations the use of fire for managing lantana is simply not appropriate.

X. Phalaris (*Phalaris aquatica*)

Existence in New South Wales

Phalaris is an introduced pasture grass which is still widely used in agriculture but can become an environmental weed in native grasslands where it outcompetes and replaces native grasses such as kangaroo grass.

Status

- Phalaris is an environmental weed of high conservation value Temperate Grasslands.
- Key threatening process: '*Invasion of native plant communities by exotic perennial grasses'* (NSW SC 2003). Activities which facilitate or exacerbate the threat of invasion by phalaris to threatened species, populations and ecological communities, and their habitats must be considered when an activity requires assessment under Part 5A of the EP&A Act.

Ecological attributes and preferred habitat

First released into Australia in 1884, it is highly invasive of native grasslands, grassy woodlands, forests, wetlands and riparian areas (Muyt 2001). It is originally from Western Africa, Southern Europe and the Mediterranean (Oram *et al. 2009)*. In 1990 it was estimated there was potentially over 3 million hectares sown. It is tolerant of frost and grows through winter and is considered to be the most productive and persistent temperature pasture grass of the tablelands and western slopes. It grows best where maximum temperatures are in the range of 15–25°C. Seeds are spread by wind. Phalaris is largely confined to the tablelands, slopes and some coastal districts (NSW Department of Agriculture 2000).

Interaction with fire

Fuel loads in grasslands dominated by phalaris are significantly higher than native grasslands and pose a high fire risk (Stoner *et al.* 2004).

Key findings and management options

Fire can be used on good quality native grassland sites where phalaris is present or nearby to keep it under control by removing material in winter and allowing native grasses to compete. On sites dominated by phalaris and where fire has been absent for long periods (i.e. 5 to 10 years), extensive chemical or mechanical follow-up will be required (G. Johnson, 2016 pers. comm.).

XI. Large-leaved privet (*Ligustrum lucidum*) and small-leaved privet (*L. sinense*)

Existence in New South Wales

Large-leaved and small-leaved privet, sometimes called broad-leaved and narrow-leaved privet and collectively referred to in this report as 'privet', are widespread weeds across the higher rainfall tablelands and coastal river valleys of New South Wales.

Status

- *NSW Biosecurity Act 2015* - Priority weeds for the Central Tablelands (Containment) and Northern Tablelands (asset Protection) regions. Weed of concern in Greater Sydney (Asset Protection), North Coast (Asset Protection) and the Riverina regions. https://www.lls.nsw.gov.au/biosecurity/weed-control

Ecological attributes and preferred habitat

Both privet species are widespread across the coastal valleys and tablelands of New South Wales, with isolated infestations further west on the slopes and plains. Both species cover an altitudinal range from sea level to over 1500 m ASL.

The first evidence of *Ligustrum* species in Australia is a record in William Macarthur's earliest catalogue and documents large-leaved privet as occurring in cultivation in south-west Sydney (Johnson 2009). Large-leaved privet was documented as being originally cultivated in 1857 at Camden Park, probably as a hedge plant. The precise introduction of small-leaved privet is less clear, but it was being distributed as a hedge plant in the early 1900s and was already well established in many temperate towns by this stage. The *Ligustrum* species were widely planted as hedges and shade trees after their introduction (Swarbrick, Timmins & Bullen 1999; Benson & McDougall 1999).

Both species of privet spread rapidly beyond the original town and farmland hedge plantings to establish within a great diversity of native vegetation communities, from coastal rainforests and riparian areas to high elevation grassy woodlands, heathlands and eucalypt forests.

Large-leaved privet is native to Japan, China and other parts of Eastern Asia (Hardin 1992; Benson & McDougall 1999), whilst small-leaved privet is native to China, Hong Kong, Taiwan, Laos and Vietnam (Hardin 1992; Johnson 2009).

Small-leaved privet is a lot more frost resistant than large-leaved privet and grows to a maximum height of 5–10 m, forming dense sheltered thickets with low light penetration, high moisture levels and limited understorey and ground layer diversity. Large-leaved privet grows to a maximum height in excess of 30 m when growing in high rainfall areas on fertile soils, a position in which it can become highly dominant within a forest structure (White, Vivian-Smith & Barnes 2009).

In optimal conditions small-leaved privet can start fruiting within about 5 years of establishment (Johnson 2009). Large-leaved privet requires a considerably longer period of development, because fruiting generally only commences when a substantial canopy has established, typically after some decades of growth (Johnson 2009). Both species bear dark fleshy fruit with a diameter of approximately 5 mm in autumn and winter. Ekert and Bucher (1999) found that seed production in large-leaved privet could be as high as 100,000–10,000,000 seeds/plant and Fox and Adamson (1986) found up to 90% germination success.

Both privet species have been recorded with an initial seed viability of up to 100% (Swarbrick, Timmins & Bullen 1999; Blood 2001; Mowatt & Smith 2004). Field studies indicate that the seed of large-leaved privet has a maximum longevity of 1–2.5 years in soil seed banks, while seeds of small-leaved privet had a viability of less than 6 months (van Aalst 1992; Panetta 2000). After being ingested by pied currawongs, seeds had a viability of 83–91% (Buchanan 1989). Bird regurgitation and defecation result in clustered seed distribution (Buchanan 1989; Mowatt & Smith 2004).

Both species are dispersed by birds, notably the pied currawong (Buchanan 1989; Bass 1989, 1990, 1995, 1996), silvereyes and various fruit pigeons and doves (White, Vivian-Smith & Barnes 2009). Both privet species are also frequently dispersed by water when occurring within the riparian zone, a landscape position that privet frequently dominates.

Bird dispersal of privet seeds from adult plants results in new infestations in surrounding landscapes (Bass 1995, 1996; Johnson 2009). Often the ground layer beneath fencelines and nearby patches of forest or paddock trees is heavily covered by privet seedlings because of their capacity to germinate and establish in low light conditions (Johnson 2009). Privet will also establish at a lower density within eucalypt forests (both Wet and Dry Sclerophyll), Grassy Woodlands, Heathlands and Forested Wetlands (Johnson 2009) in close proximity to seed sources. In optimal conditions these seedlings will develop into substantial thickets and alter the structure of native vegetation communities with a consequent reduction in native plant diversity. This process has been responsible for significant changes in many fertile coastal and tablelands landscapes across New South Wales, with many hillsides and stream banks covered in dense thickets of small-leaved privet or dense forests of large-leaved privet. This process is ongoing with privet stands establishing and ‘creeping’ across the landscape and detrimentally impacting on native plant diversity.

The establishment of thickets and forests of privet alters the flammability of the landscape. Privet infestations create shifts in vegetation from previously open and grassy habitats to dense habitats that retain higher levels of moisture for longer periods. This strongly influences the fire regime and has the potential to, in the most severe of instances, lead to permanent fire exclusion from previously frequently-burnt habitats.

Interactions with fire

The foliage of both privet species is vulnerable to being scorched and killed by fire (Swarbrick, Timmins & Bullen 1999; Batcher 2000; Munger 2003), however, both species sucker prolifically following disturbance, a trait that contributed to the widespread use of both species of privet as hedges in suburban gardens and on farms (see Johnson 2009).

Intense fire in the ground and shrub strata will kill most seedlings and some stems, but significant resprouting and suckering of larger stems and trunks occurs following even high intensity wildfires (Munger 2003). Large-leaved privet is particularly susceptible to fire because of its thin bark and a lack of other fire-protective features (Swarbrick, Timmins & Bullen 1999).

Johnson (2009) suggests that the main difficulties in using fire within eucalypt-dominated woodlands and forests that have been invaded by privet is the absence of perennial grasses such as kangaroo grass due to being shaded out by privet (see section above for a description of the shifts in fire regime caused by privet invasion and establishment).

In Toowoomba during successive dry winters in El Niño periods, wilting and leaf drop from large-leaved privet onto exposed slopes and ridges, increased the fuel load resulting in a hot fires (Swarbrick, Timmins & Bullen 1999). Swarbrick, Timmins & Bullen (1999) found that most saplings and small trees under 5 cm diameter were killed by the fires. They also found that larger trees were killed, but in a number of cases plants regenerated from the unburnt stem material, requiring follow-up herbicide application to achieve better control outcomes. Noting a lack of post-fire recruitment, Swarbrick, Timmins & Bullen (1999) suggested that these hot fires killed much of the weed seed bank at and near the soil surface.

In the southern USA repeated annual ‘cool’ burns have eliminated large-leaved and small-leaved privet from a range of habitat types (Munger 2003). In Australia relatively frequent fires are probably essential to control seedlings and prevent the establishment of privet within sclerophyll forests and woodlands (Swarbrick, Timmins & Bullen 1999).

Key findings and management options

- Generally fire is not a suitable tool for managing privet because of its frequent occurrence within sensitive rainforest habitats and riparian areas.
- The use of relatively frequent fire (within recommended and regulated thresholds) in grassy eucalypt forests and woodlands is recommended to kill invading privet seedlings and to prevent the establishment of privet thickets or forests.
- If a wildfire has occurred within an area infested with privet then post-fire follow-up with a range of control techniques such as herbicide stem injection (frilling), foliar spraying and manual or mechanical removal is strongly recommended.

XII. Scotch broom (*Cytisus scoparius* subsp. *scoparius*)

Existence in New South Wales

Scotch broom is a large invasive perennial shrub of the cool temperature areas of the tablelands from Glen Innes to Victoria. It is also known as English broom. It rapidly colonises disturbed areas, where it forms large dense thickets, but it is also capable of invading relatively intact areas of native vegetation. (AWC, 2012b)

Status

- *NSW Biosecurity Act 2015* - Listed as a State-wide priority weed (Asset Protection). Priority weed for the Central Tablelands (Asset Protection), Greater Sydney (Asset Protection), Hunter (Containment), Murray (Eradication), North Coast (Eradication), North West (Containment), Northern Tablelands (Containment), Riverina (Eradication except for Snow Mountains Council LGA - Containment) and South East (Containment) regions. https://www.lls.nsw.gov.au/biosecurity/weed-control
- Weed of National Significance (Thorp & Lynch 2000).
- Key threatening process: *Invasion and establishment of scotch broom (Cytisus scoparius)* (NSW SC 2007). Activities which facilitate or exacerbate the threat of invasion by Scotch broom to threatened species, populations and ecological communities, and their habitats must be considered when an activity requires assessment under Part 5A of the EP&A Act.

Ecological attributes and preferred habitat

Introduced from Western Europe, it was established in Sydney by 1803 (OEH 2014). By 1900 it had become a major invasive weed of grasslands and open woodlands in altitudes above 600 m ASL.

Broom can grow to 4 m in height, but after 10 years plants begin to lose rigidity and fall over, especially after heavy snowfalls (Smith 1994). Broom is frost tolerant and can survive periods of drought. It is also a nitrogen fixer so it alters soil conditions to favour more broom over competing native species. It may also be toxic to grazing animals (AWC, 2012b). It first flowers and fruits after two years mainly in spring–summer but can continue all year in warm and wet conditions. Plants are spread by large quantities of seed released from an explosive pod. Most seed falls within a few metres of the parent plant. Some further dispersal may occur via ants, animals, machinery and floods. Once it is established it can change the existing fire regime, outcompete native vegetation and harbour feral animals such as foxes, pigs *(Sus scrofa)* and rabbits (AWC, 2012b). Seeds buried by ants can survive cool fires and germinate later if the soil is disturbed or a hot wildfire occurs. The seeds are hard and only small amounts germinate at one time. This combined with seed longevity means large soil seed banks of 60,000 seeds/m^2 may be present under mature shrubs. Seeds can be buried up to 5 cm deep and retain viability for 30 years (Hoskins, Sheppard & Smith 2000) The WONS Brooms Strategic Plan 2014–2017 reports viability up to 80 years (AWC, 2012b) but does not identify the source.

Interaction with fire

Downey and Smith (2000) found that seed germination under adult plants reduces over time and older plants produce less seeds in the absence of disturbance such as fire. The dense infestations will self-thin. This allows some native grasses and herbs to establish, and the less-dense stands are less likely to harbour feral pigs. If fire is used to kill mature plants, or remove senescent material, mass germination of seeds occurs and denser, more vigorous stands become established and quickly produce large amounts of seed again. They recommend that although fire can be used to stimulate seed germination for chemical treatment and depleting the soil seed bank. Unless resources are available for follow-up treatments over many years, a denser infestation is likely to result.

The *Weeds of National Significance broom management manual* (OEH 2014) advises that fire should only be used as part of an integrated weed control plan with other treatments. It may be difficult to introduce fire of sufficient heat due to the lack of fine fuel below mature plants and high moisture levels. In dry conditions broom burns readily and can increase the intensity and spread of wildfires. Plants should be cut and spread out to dry out well before using fire. Spot spraying of any regrowth in the months before fire is applied will ensure a higher intensity burn. Follow-up every few months, especially after rainfall, will need to be ongoing for many years until the soil seed bank is depleted.

Key findings and management options

- A hot fire will kill mature plants and flush out deeply buried seeds but should only be used where sufficient resources are available to conduct intensive follow-up treatments on an ongoing basis, as large quantities of seed may be dormant for 20–30 years. A denser infestation will result without sufficient follow-up treatments.
- Spot spraying of regrowth in the months before applying fire to encourage the growth of native grasses and dry the ground fuel results in a higher intensity burn.

XIII. Serrated tussock (*Nassella trichotoma*)

Existence in New South Wales

Serrated tussock is an invasive tussock-forming perennial grass of grassy woodlands and native grasslands of temperate areas from the Northern Tablelands to the Southern Tablelands and South Coast.

Status

- *NSW Biosecurity Act 2015* - Listed as a State-wide priority weed (Asset Protection). Priority weeds for the Central Tablelands (Asset Protection), Central West (Containment), Greater Sydney (Containment), Hunter (Prevention), Murray (Eradication) North West (Containment), Northern Tablelands (Containment), Riverina (Eradication) and South East (Containment) regions. Watch list for North Coast region. https://www.lls.nsw.gov.au/biosecurity/weed-control
- Weed of National Significance (Thorp & Lynch 2000).

Key threatening process: *Invasion of native plant communities by exotic perennial grasses* (NSW SC 2003). Activities which facilitate or exacerbate the threat of invasion by serrated tussock to threatened species, populations and ecological communities, and their habitats must be considered when an activity requires assessment under Part 5A of the EP&A Act.

Ecological attributes and preferred habitat

Serrated tussock is a summer-growing grass which can live for 20 years. Seeds are spread by wind and can germinate and compete strongly in low through to high fertility sites. Seed mostly germinate in autumn but can germinate any time of year after rain. The majority of seed germinate within 3 years but in controlled conditions some were still viable after 20 years (NSW DPI 2015c).

Interaction with fire

A hot fire will not kill established tussocks but it will kill surface seed and stimulate seed germination. So follow-up control treatments are needed to deplete the soil seed bank (Osmond *et al.* 2008). Muyt (2001) records that fire can be used to burn away dried foliage to allow better penetration of herbicides and to identify individual tussocks when mixed in with desirable species. Burning in spring before flowering will restrict seed development for that season. Fire can be used to kill seeds present on the soil surface (Muyt 2001). The CRC weed management guide (CRC for Australian Weed Management 2003) states that annual burning can help control serrated tussock in areas of low rainfall and low soil fertility.

Key findings and management options

- A hot fire will not kill established tussocks but can be used to kill surface seed and stimulate below-ground seed-germination so when follow-up controls are undertaken, the soil seed bank is depleted
- Burning in spring before flowering will restrict seed development for that season.
- In areas of low rainfall and low soil fertility annual burning can help control serrated tussock.

XIV. South African pigeon grass (*Setaria sphacelata* var. *sericea*)

Existence in New South Wales

South African pigeon grass, hereafter setaria, is widespread in the higher rainfall and wetter parts of the subtropical coastal river valleys of New South Wales, north from the Illawarra, but mostly north of the Manning River.

Status

- Environmental weed of recovering North Coast grassy ecosystems

Ecological attributes and preferred habitat

Setaria is widespread across the floodplains and lower slopes of the coastal river valleys of the NSW North Coast. It has spread rapidly from pastures into a range of subtropical native vegetation communities. Some varieties are frost-hardy which has expanded the range westwards up the coastal river valleys and into cooler and more frost-prone landscapes.

Setaria is a tall tropical grass growing to a height of approximately 2 m. It forms a dense sward with a high biomass contributing a high fuel load when conditions are suitable. It is native to a significant part of the African continent, ranging from the tropical basins of Central Africa to South Africa. Setaria grows well on a wide range of soils from clay to sands and waterlogged organic soils. It tolerates acidic soils and moderate levels of waterlogging. Setaria has a greater tolerance to cool temperatures than most other tropical grasses (NSW DPI 2000).

Intentional introductions of several cultivars of setaria for pasture production on coastal floodplains began in the 1960s and have continued until the present day (NSW DPI 2000, 2004b). Setaria has spread rapidly beyond improved pastures and has established across a wide range of landscapes, from inundated floodplains to slopes on sandstone ridges. Setaria has established and spread within a diversity of vegetation communities, with the most significant infestations within coastal floodplain wetlands and forests (Coward 2015) and the margins of coastal rainforests (NCC Upper Coldstream Biodiversity Project, unpublished data). Seeds are spread by wind, water and gravity. Heavy recruitment has been noted in recently burnt sites in the coastal parts of the Clarence Valley (M. Graham 2012–2016, pers. obs.).

Setaria establishes a tall, dense perennial sward that can completely exclude and replace diverse native understorey assemblages. This is particularly the case within forested wetlands, the margins of freshwater wetlands (both threatened ecological communities) and grassy woodlands on the coastal plain (NCC Upper Coldstream Biodiversity Project, unpublished data) where it not only replaces the native understorey, but suppresses regeneration of canopy species (Coward 2015).

Interactions with fire

In an experimental manipulation of grassland cover within Serengeti National Park, Tanzania, using grazing and fire to manage and manipulate grass cover, Belsky (1992) found that fire increased the cover of setaria after grazing had previously reduced cover. Setaria is native to the Serengeti and its response to the various experimental disturbances is occurring within its original intact and functional ecosystem within which it is exposed to the full range of predators, pathogens and other mediating processes. In Australian ecosystems setaria is free of most predators and pathogens and, when pastures are rank, is rarely grazed on by domestic stock (NSW DPI 2004b).

On the NSW North Coast where many native vegetation communities on floodplains have been invaded by setaria, fire has been found to promote the spread of the species. In particular, post-fire germination of the species has contributed to the establishment of new infestations. In these landscapes fire has also caused rapid post-fire regeneration of existing patches and assisted in their expansion. In the coastal floodplains of the Clarence Valley this has ultimately increased the cover and density of setaria in the landscape (NCC Upper Coldstream Biodiversity Project, unpublished data).

Coward (2015) details a successful case study where the use of herbicide to kill setaria was integrated with the subsequent use of fire to successfully restore a Forested Wetland community (a threatened ecological community) near Evans Head. Treatments ranged from herbicide application only, to herbicide application followed by a cool burn, and herbicide application followed by a hot burn. After herbicide application, the tall dense cover of setaria died leaving a large standing mass of fuel which burnt with a moderate to high intensity across a significant proportion of the site. The quadrats that experienced a hot fire had achieved over 50% native cover within 5 months, while the unburnt area achieved only half (25%) that cover. Both treatments achieved a similar recovery of natives over time, but much higher spot spraying inputs were required over longer time frames in the unburnt areas compared to the hotter burn areas (Coward 2015).

Regeneration of setaria and ragweed (*Ambrosia* sp.) was prolific following the burn, with little initial native regeneration. Near-complete herbicide spraying was required to control the initial germination of weeds, after which extensive native regeneration began with limited weed recruitment which required limited spot spraying (Coward 2015). Over the course of 3 years, 35 species of groundcover established (from the soil seed bank) and seven tree and shrub species established (from seed fall from nearby intact vegetation). This happened on a site where previously a near-complete monoculture of setaria existed (Coward 2015). This project has provided valuable insights into the benefits of integrating fire with more conventional bushland regeneration techniques, such as herbicide application, to achieve good regeneration outcomes. These insights have been applied within the Minyumai Indigenous Protected Area which has commenced a similar project to restore forested wetlands within a setaria-dominated floodplain landscape approximately 5 km south of the Coward property.

Key findings and management options

- Fire should not be used in isolation to manage setaria as it is likely to promote growth and exacerbate existing infestations.
- Using a strategy that integrates herbicide application with fire and follow-up regeneration work is recommended for restoring floodplain habitats degraded by setaria. These methods may also be valuable in restoring other ecosystems degraded by setaria, such as grassy Wet Sclerophyll Forests on slopes, and there are numerous opportunities for trialling and perfecting such treatment approaches across the coastal river valleys of the North Coast.

9. Conclusions

The native vegetation communities of New South Wales contain a great diversity of native species, but many weed species have invaded and degraded these vegetation communities. This review has investigated the interaction of fire with a range of significant weed species in the native vegetation of the State. The review has found that the interactions of fire with weeds are complex, varied and often difficult to predict. In many instances a lack of funding and other resources hampers the achievement of effective and lasting ecological restoration outcomes, and this can also hinder new research projects. Another factor preventing the achievement of good weed and fire management outcomes is that there is a general lack of specific knowledge of the interaction of fire with most of the weeds that are degrading the native vegetation of New South Wales.

In many instances fire was found to exacerbate existing weed infestations, creating additional ecological degradation and leading to a reduction in biodiversity. Fire was found to have a positive influence on the restoration of degraded ecosystems for a relatively limited subset of the weeds investigated. Fire is rarely, if ever, a ‘silver bullet’ solution for weeds within native vegetation and in almost all instances reviewed, combinations of fire with other weed management techniques such as mechanical control or herbicide application are required to achieve good and lasting ecological restoration outcomes.

It is the aim of the Hotspots team that the information presented in this document will be used as a platform for further discussion between fire and weed management agencies, practitioners and the wider community. Our aim is to incorporate new knowledge and experience as it is gained.

A summary of the key findings and management options for weed species considered as part of this review can be found in Table 1.

Table 1. Summary of weed species reviews and management options

Weed species	Regions of NSW where found	Value of fire as management tool – key findings	Management options
African Lovegrass (*Eragrostis curvula*)	Can be found on roadsides throughout NSW but is mainly a problem on the **tablelands**, **Western Sydney** and the **South Coast**	Early results of a long-term trial at Cattai and Scheyville national parks on the Cumberland Plain, west of Sydney, found that using a combination of fire and spraying with the partially selective grass herbicide, Flupropanate, was effective in reducing the cover of African lovegrass (Sanders *et al.* 2016). In the Bega Valley on the South Coast, the use of fire and Flupropanate is not recommended due to the potential impacts on non-target native grass species such as weeping grass (*Microlaena stipoides*) and the likelihood that areas are left as bare ground for long periods. This leaves the soil vulnerable to erosion and provides conditions most favoured by African lovegrass (J. Dorrough 2016, pers. comm.).	**Wildfire** – An integrated approach may include sowing of native pasture species to help restoration (NSW DPI 2014a). **Prescribed fire** – In higher rainfall areas, burning African lovegrass to reduce the sward density, stimulate native plant species and allow more targeted herbicide control can be effective (NSW DPI 2014a). In areas of low to moderate rainfall where the soil may be exposed for long periods, the use of fire is less effective as replacement grass species take longer to establish.

African olive (*Olea europaea* subsp. *cuspidata*)	Mainly located in the **Central Coast**, west of Sydney, and the **South Coast,** but spreading to the North Coast and western slopes	Fire will not kill mature African olive trees (Spennemann 1998). Large mature trees have been known to resprout 18 months after fire (Cuneo & Leishman 2006). Fire will stimulate seed germination from the soil seed bank and kill young trees.	**Wildfire** – Follow-up treatment to kill seedlings which will germinate after the fire. **Prescribed fire** – Fire is not effective as a control method for mature trees but can be used to stimulate the germination of seeds in the soil seed bank which can be killed by a low intensity burn or other control methods**.**
Bitou bush (*Chrysanthemoides monilifera* subsp. *rotundata*)	Most severe on **Central** and **North Coast** but also occurs sporadically on the **South Coast**	Hot fire can kill mature plants but they will readily resprout in less intense burns (Thomas *et al.* 2006). Hot fire can kill seeds in the upper part of the soil seed bank (Thomas *et al.* 2006). Fire will stimulate the germination of seeds in the seed soil bank which can be viable up to 10 years (DEC 2006).	**Wildfire** – Follow-up control of mechanical or chemical treatment (Lindenmayer *et al.* 2015). **Prescribed fire** – High intensity fire can be used to kill seeds, seedlings and mature plants and exhaust the soil seed bank if followed up with ongoing mechanical or chemical treatment (Vranjic *et al.* 2012).

Blackberry (*Rubus fruticosus*) species aggregate	Can be found in most regions in the east of the State but is more abundant in cooler areas like the **tablelands** and **South Coast**	Burning can kill seasonal canes but will not kill the root crowns. Fire can be used to increase the accessibility of blackberry infestations for follow-up treatment (NSW DPI 2014c). Fire will stimulate native seed germination under and around blackberry thickets.	**Wildfire** – Can provide opportunities for follow-up herbicide control. Seeding with native shrubs may help restore sites where the native soil seed bank is depleted. **Prescribed fire** – Can be useful to reduce the density of large thickets and stimulate seed germination for other control methods (NSW DPI 2009).
Boneseed (*Chrysanthemoides monilifera* subsp. *monilifera*)	It is naturalised in coastal districts from the **Hunter River** to **Moruya** on the South Coast	Boneseed does not burn well under normal growing conditions, but a hot fire can kill adult plants and stimulate seed germination for other control methods to deplete the seed soil bank.	**Wildfire** – Can provide opportunities for follow-up herbicide control. **Prescribed Fire** – Fire of at least moderate intensity can be used to kill mature and young plants. Cutting large individuals and spreading them out 12 months before a prescribed burn will increase the intensity and effectiveness of a burn (Brougham, Cherry & Downey 2006).

Camphor laurel (*Cinnamomum camphora*)	Mainly found in coastal and hinterland areas from the **North Coast** to **Central Coast** and the **Blue Mountains**	Fire is not appropriate in rainforest habitats and has limited value in wet sclerophyll habitats. Camphor laurel will usually resprout after fire	**Wildfire** – Post-fire control should be undertaken using appropriate techniques, such as foliar spraying and stem injection, to prevent rapid establishment and dominance of the habitat by camphor laurel post-fire. **Prescribed fire** – Can be used to kill seedlings in drier plant communities, especially grassy habitats.
Chilean needle grass (*Nassella neesiana*)	Mainly found in **Northern Tablelands** and **north-west slopes** and the **Southern Tablelands** and **south-west slopes**	Fire can be used as part of an integrated weed action to prevent seed setting, to burn off standing seed and stimulate the growth of seeds in the soil seed bank. See the *National best practice management manual for Chilean needle grass* (Vic. DPI 2007) Fire will stimulate the germination of native grassland species. Follow-up herbicide spraying of Chilean needle grass will assist native species to compete.	**Wildfire** – May provide opportunity to introduce competing plant species. **Prescribed fire** – At sites where summer-growing native grasses are present, burning in spring can assist native species to compete with the winter growing Chilean needle grass and reduce the resprouting success of mature plants. Follow-up control of new seedlings will be required as seeds that are not killed by fire will germinate later. Sowing gaps with native grasses will assist recovery (Vic. DPI 2007).
Coolatai grass (*Hyparrhenia hirta*)	Common in **northern NSW** and increasing in the south	This species tolerates frequent fire well and will out-compete native grasses post-fire (McCormick, Lodge & McGufficke 2002).	**Wildfire** – May provide opportunity to introduce competing plant species. **Prescribed fire** – Not recommended.

Lantana (*Lantana camara*)	Mostly east of the Great Dividing Range from **Eden to the Qld border**	A hot fire can kill mature plants and seedlings, but plants will often resprout from basal dormant buds (Swarbrick, Timmins & Bullen 1998). Fire is a significant disturbance that facilitates the invasion of lantana into a site (Gentle & Duggin 1997b).	**Wildfire** – May provide conditions for appropriate chemical treatment. **Prescribed fire** – In dry sclerophyll and grassy woodland habitats where frequent fire is more appropriate than in moist habitats, fire can slow down the infestation of lantana into new areas. In some areas, pre-burn herbicide treatment of dense thickets may allow the use of fire to reduce biomass and encourage regeneration of competing species. Follow-up treatment of seedlings and resprouting of old plants will be ongoing.
Phalaris (*Phalaris aquatica*)	Environmental weed of native grasslands on the **Tablelands**	Fuel loads in grasslands dominated by phalaris are significantly higher than native grasslands and pose a high fire risk (Stoner *et al.* 2005).	**Wildfire** – May provide opportunity to introduce competing plant species. **Prescribed fire** – Can be used on good quality native grassland sites where phalaris is present or nearby, to keep it under control by removing material in winter and allowing native grasses to compete. On sites dominated by phalaris and where fire has been absent for long periods (5–10 years), extensive chemical or mechanical follow-up will be required (G. Johnson 2015, pers. comm.).

Privet Large-leaved privet (*Ligustrum lucidum*) Small-leaved privet (*Ligustrum sinense*)	Both privet species are widespread across the **coastal valleys and Tablelands of NSW**	The foliage of both privet species is vulnerable to being scorched and killed by fire (Swarbrick, Timmins & Bullen 1999; Batcher 2000; Munger 2003), however both species sucker prolifically and are quick to recover. Both species, particularly broad-leaved, tend to occur in moister situations where fire is infrequent and may be inappropriate or difficult to introduce.	**Wildfire** – If a wildfire has occurred within an area infested with privet, then post-fire follow-up with a range of control techniques such as herbicide stem injection (frilling), foliar spraying and manual or mechanical removal is strongly recommended. **Prescribed fire** – Generally fire is not a suitable tool for managing privet because of its frequent occurrence within sensitive rainforest habitats and riparian areas. The use of relatively frequent fire within grassy eucalypt forests and woodlands is recommended to kill invading privet seedlings and to prevent the establishment of privet thickets or forests.
Scotch broom (*Cytisus scoparius* subsp. *scoparius*)	Generally in the **cooler high altitude** areas of the State. Dense infestations at **Barrington Tops** and near **Braidwood**	If fire is used to kill mature plants or remove senescent material, a mass germination of seeds is produced and denser stands become established quickly producing large amounts of seed again. Fire can be used to stimulate germination and deplete the soil seed bank if resources are available to apply follow-up treatments over many years (Downey & Smith 2000).	**Wildfire** – May provide conditions for appropriate chemical treatment of emerging seedlings. **Prescribed fire** – Fire should only be used where sufficient resources are available to conduct intensive follow-up treatment on an ongoing basis, as large quantities of seed may be dormant for 20–30 years. A denser infestation will become established without adequate follow-up treatments.

Serrated tussock (*Nassella trichotoma*)	Occurs from the **Northern Tablelands** to the **Southern Tablelands** and **South Coast**	A hot fire will not kill established tussocks but it will kill surface seed and stimulate seed germination, so if follow-up controls are undertaken it can be used to deplete the soil seed bank (Osmond *et al.* 2008).	**Wildfire** – May provide conditions for appropriate chemical treatment of emerging seedlings. **Prescribed fire** – Fire can be used to kill surface seed and stimulate seed germination so if follow-up controls are undertaken it can be used to deplete the soil seed bank. It can also be used to increase the effectiveness of chemical and mechanical control.
South African pigeon grass *(Setaria sphacelata* var. *sericea*)	Found from South to North Coast but mainly a problem of the **floodplains and coastal river valleys of northern NSW**	Fire can increase the cover and density of setaria in the landscape (NCC Upper Coldstream Biodiversity Project, unpublished data). Fire should not be used in isolation to manage setaria as it is likely to promote growth and exacerbate existing infestations.	Using a strategy that integrates herbicide application with fire and follow-up regeneration work is recommended for restoring floodplain habitats degraded by setaria.

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