Weed management for the land-scape scale restoration of global temperate grasslands: a review.

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Abstract

Globally, temperate grasslands have been significantly altered and subsequently degraded as a result of increased human population, urbanisation, and agriculture. Weeds now dominate most of these ecosystems, resulting in the loss of ecosystem services, reduced carrying capacity for farmers, and loss of habitat for native animals. This paper reviews the literature of temperate grassland restoration efforts from across the globe, and observes what techniques and combinations have been used successfully and unsuccessfully to reduce weed dominance and promote native recruitment and establishment. The findings of this review clarify that weed management should be ongoing in all projects, while optimal revegetation methods and grazing regimes are specific to site location and study scope. There is a need for an increase in long-term monitoring of restoration projects in order to make assumptions with greater confidence.

Introduction

Temperate grasslands once covered almost 9 million km^2, which is equivalent to about 8% of the earth's surface (IUCN, 2013). They include the Prairies of North America, the Pampas of South America, the South African Veldts, the Tussock grasslands of Australia and New Zealand, and the Steppes of Eurasia (Table 1). These biomes are often species rich (Faber-Langendoen & Josse, 2010), providing natural habitat for many plants, animals and soil biota. In addition, these grasslands offer invaluable ecosystem services such as high-quality forage for herbivores (Boval & Dixon, 2012), harbour pollinators for crops and native species (Bendel *et al* ., 2019), provide significant levels of carbon sequestration (Eze*et al* ., 2018), and are places for many recreational and cultural activities (Gomez-Limon & de Lucio, 1995). They also afford many other environment stabilizing services, such as soil erosion control and mitigation of flood waters (Sankaran & Anderson, 2009).

Given the significance and contribution of this ecosystem, they're currently one of the most altered ecosystems in the world (Suttie*et al* ., 2005), warranting immediate action to restore these beneficial services. Estimates suggest that 70% of these ecosystems were altered or degraded before 1950, and a further 14% by 1990 (Hassan*et al* ., 2005). This decline in ecosystem health is directly attributed to rapid population growth and subsequent urban expansion (Williams *et al* ., 2005), as well as the concomitant conversion of these fertile ecological systems into sites for agriculture, particularly for cropping systems and livestock grazing (Martin *et al* ., 2005; Prober *et al* ., 2005; Bartolome *et al* ., 2009; Sankaran & Anderson, 2009). While increased protection of the remaining intact system is critical, it is not enough to ensure the future resilience and functionality of these systems. Therefore, this paper reviews restoration methods, both active and passive, that reduce invasive plant biomass within global degraded temperate grasslands to promote the return of natives and subsequent ecosystem functionality at a landscape scale.

Ecological Restoration Methods

Ecosystem degradation is the movement of a high functioning and healthy ecosystem into an altered state whereby ecosystem functions are reduced or lost as a result of a single or multiple disturbance event, usually due to human actions (Suding & Hobbs, 2009). Healthy ecosystems are often resilient enough to withstand moderate disturbances, and these events can be important for maintain biodiversity and healthy ecosystems. When the intensity and/or frequency of these disturbance events change, an ecosystem can undergo hysteresis and pass through irreversible degradation thresholds (Suding & Hobbs, 2009). An example of this was observed in the grasslands of California where the combined factors of weed invasion by Mediterranean species and altered grazing regimes from the introduction of livestock as well as multiple severe drought seasons resulted in the sever degradation of this ecosystem, pushing it into an alternative stable state (Bartolome *et al* ., 2009). Different levels of degradation can influence the scale of restoration required, depending on if the biotic factors (weed invasion), abiotic factors (drought or altered fire regimes) or a combination of both are affected.

Throughout the world, grasslands are currently subjected to multiple degrading pressures. The most common of these pressures include; habitat fragmentation, which has been observed in Australia (Prober *et al* ., 2005), New Zealand (Standish *et al* ., 2009) and Europe (Kiehl, 2010), altered grazing pressures in the USA (Martin *et al* ., 2005; Bartolome *et al* ., 2009), Africa (Sankaran & Anderson, 2009) and Australia (Prober *et al* ., 2005), desertification and bush encroachment have been observed in Africa (Sankaran & Anderson, 2009), and climate change has been linked to the gradual decline in grasslands throughout China (Zha & Gao, 2011). These degrading pressures often act to promote the invasion of exotic plants, which in turn create positive feedback loops that maintain the degraded altered state. The alternative state theory explains how internal disturbances and external shocks lead to positive feedback loops which promote a stable degraded state (Chisholm *et al* ., 2015). In this state, the degrading factors have altered the environment to promote their own development, as observed in south east Australian grasslands where annual exotic grasses outcompete the perennial native grass, Themeda by developing new positive feedback loops that increases soil nitrogen, and unless the available soil nitrogen levels are reduced, the invasive species will maintain its competitive edge (Prober *et al* ., 2009). Further, cross-facilitation of invasive plant feedback loops has been identified by observing *Agropyron cristatum* in northern USA, which alters native soil biota and thus, soil dynamics (Jordan *et al* ., 2008). This reduces the competitiveness of the native vegetation, promoting niche availability for the invasive*Bromus inermis* (Jordan *et al* ., 2008). It is in this aggressive context that ecological restoration needs to reverse and prevent further degradation, and then assist in the recovery of an ecosystem. An important element in this task is to more clearly understand how human behaviour influences different aspects of an ecosystem (Schroder, 2009). Restoration models have been evolving for several decades (Suding & Hobbs, 2009), and have proven to be successful for prioritising large- and small-scale restoration projects.

The restoration of weed-dominated grasslands has received extensive both popular and academic interests. Native tufted perennial grasses have been described as keystone species as they resist weed invasion and maintain ecosystem processes (Stromberg *et al* ., 2009; Prober*et al* ., 2005). Human induced disturbance has resulted to the decline of these native grasses and thus non-native perennial grasses have established, which significantly reduces the carrying capacity and biodiversity. This has been observed in grasslands dominated by*Nassella trichotoma* throughout south-eastern Australia (Campbell & Nicol, 1999; Jacobs & Everitt, 2012), South Africa (Joubert, 1984) and New Zealand (Lameroux *et al* ., 2011; Lusk *et al* ., 2017). Annual weeds are also highly problematic, and outcompete native perennial grasses in their early life stages, and this is most prevalent after disturbance (Musil *et al* ., 2005; Bartolome *et al* ., 2009; James *et al* ., 2011). Because invasive plants are generally one of the main drivers for holding these ecosystems in degraded states, the main focus of restoration efforts is on reducing the dominant weed population and promoting competition from native species. Passive and active restoration techniques have been used in diverse combinations to achieve this outcome at varying levels of success throughout different temperate grasslands (Table 2).

Passive restoration

Passive ecological restoration involves removing human induced degrading pressures from a site with minimal remediation. In many cases, it is presumed that non-target species will expand without human intervention, however many passive restorations have observed weeds decline with sufficient native recruitment (Sinkins & Otfinowski, 2012; Valko*et al* ., 2017). Notable vegetation shifts often occur within 10 years of rest (Smallbone, 2014), however, some may take several decades to reach similar species richness as the remnant sites, and even then, the composition of vegetation can significantly differ (van de Merwe & van Rooyen, 2011).

Successful passive restoration involves careful grazing management and the ability for target species to recruit and establish. Rapid recovery of a degraded Hungarian alkaline grassland was observed (within one year) in sites directly adjacent a natural grassland, and within six years, all sites where restored regardless of proximity to the remnant site (Valko *et al* ., 2017). In this example, the dispersal of native plant propagules was promoted by livestock roaming between natural and degraded sites (Valkko *et al* ., 2017). Grazing animals were also observed to provide an important service in maintaining species richness for highly productive Themeda grassland in south-east Australia (Schultz *et al* ., 2011). Grasslands often require disturbances such as grazing or fire to maintain species richness and grazing animals remove excess phytomass in order to generate niche space for rarer species.

While grazing plays an important role in maintaining highly productive grasslands, those suffering extensive degradation or of lower productivity often benefit from the complete removal of grazing livestock. Grazing exclusion is a cost-effective tool for passive restoration, particularly if native species are well represented. A long-term (20 and 30 years) grazing exclusion zone was developed in the steppe grasslands of China, which observed an increase in perennial grass cover, as well as higher density bud banks of these favourable grasses when compared to the grazing sites (Zhao *et al* ., 2019). The long-term (40 years) removal of cattle from northern fescue prairies in Canada was effective for reducing some invasive plants, but not others, including *Poa pratensis* , which in some areas occupied up to 90% of the canopy (Sinkins & Otfinowski, 2012). Generally, grassland species have short-lived seedbanks and if desirable species are rare, or the site is isolated from remnant patches, the seedbank will continue to diminish (Bossuyt & Hermy, 2003). Further, the recruitment of native species in degraded temperate grasslands is rare, as seedlings often fail to survive as a result of competitive weed interactions (Morgan, 2001; Lenz & Facelli, 2005). This demonstrates that isolated sites dominated by aggressive weeds may not be suitable for passive restoration. While passive restoration has proven successful under specific conditions, in sites where invasive plants have dominated for several decades and biotic and abiotic thresholds have been crossed, active intervention will be required.

Active restoration

Active ecological restoration involves the integration of management techniques, such as revegetation, herbicide application, or mechanical soil disturbance to take an ecosystem from a degraded state to one that is functional, self-sustaining and resilient. In weed dominant systems, restoration efforts that focus to remove invasive plants and promote dense, native competition are often the most successful. In order to actively restore a degraded landscape, understanding the sites history can be critical. The history of a site can identify the factors that moved it into a degraded state, and whether these changes occurred rapidly or continuously over an extended interval (Prober & Thiele, 2005). Further, the historic vegetation cover can act as a restoration target, and guide managers on revegetation assemblages (Prober & Thiele, 2005). Active restoration of weed dominated temperate grasslands should consider; i) the removal of the weeds biomass, ii) manipulation of the soil to return it to remnant condition, iii) revegetation of native propagules, and ix) site specific grazing management.

Targeting weed biomass

The removal of dense weed biomass is critical for reducing competition for naturally recruiting native's species, or those added via revegetation efforts. Weeds are often fast growing and form dense canopies, which reduces light to the soil and can thus restricts the germination and subsequent growth of native seeds or seedlings. Further, many annual grassland weeds have higher nutrient requirements than native perennial grasses, thus creating a highly competitive environment for natives to establish. The most commonly used methods for reducing weed biomass include hand removal, herbicide application, and fire.

Grubbing, or hand weeding, is a restoration technique that completely removes unwanted plants (Tikka *et al* ., 2001). While highly effective, this method is also very labour intensive, and is usually only appropriate for smaller scale projects (Gibson-Roy *et al* ., 2007). That said, every three years, community efforts have successfully removed 34% of the invasive perennial grass, *N. trichotoma*throughout Canterbury, New Zealand, which has contained the population from further expansion (Bourdot & Saville, 2019). Grubbing is the best solution for sites where weeds are newly emerging and easy to remove, or where only a few individuals have established, such as roadsides. Grubbing can prove a critical tool for post restoration management by quickly removing reinvading weeds. Grubbing is one of the most effective methods to reduce competition for space, light and soil nutrients as the whole plant is instantly removed.

Herbicide application is often an economically viable and effective solution for reducing weed competition. Herbicide works most effectively when integrated with other treatments, as seen by Johnson *et al* . (2018) who observed spot-spraying weeds with glyphosate significantly improved the establishment of native forbs seeds when combined with fencing, and the removal of leaf litter. Aerially spraying clopyralid (at a rate of 37.4 L/ha) was successful in reducing woody weed encroachment and enhancing plant diversity when combined with prescribed burning (Ansley & Casellano, 2006). Waller *et al* . (2016) also observed significantly improved native establishment when herbicide was combined with fire, tillage and rodent exclusions. In a degraded grazing exclusion zone, Huddleson & Young (2005) identified herbicide application on its own was effective for not only reduced annual weed competition by 40%, but increased native establishment ten-fold.

In some cases, herbicide application was ineffective at improving native establishment (Cole *et al* ., 2005; Conrad & Tischew, 2011). Spot-spraying Snapshot (a pre-emergent herbicide containing trifluralin and isoxaban) at 2.5kg per 100m^2 significantly reduced the emergence of native forbs compared to the controls in South Africa (Musil *et al* ., 2005), however was effective for controlling invasive annual grasses. In New Zealand, boom-spraying flupropanate at 1.49kg a.i/ha reduced native pasture grass by 89% (Lusk *et al* ., 2017). In these cases, using herbicides selectively can enhance restoration outcomes. Selective herbicides are used to kill the unwanted weeds, while the desirable species remain unharmed, and this can be attributed to; plant chemistry, physical growth parameters and plant physiology (Sutton, 1967). It is important to note that the constant use of herbicides within an ecosystem can promote the emergence of herbicide resistant populations, thus reducing its long-term effectiveness. Resistance to arguably the world's most important herbicide, glyphosate, has already been observed in several weeds (Powles 2008), including Conyza spp. (Feng *et al* ., 2004; Urbano *et al* ., 2007) and Lolium spp. (Baerson *et al* ., 2002; Yanniccari *et al* ., 2017). It is considered important, therefore, that herbicides should be used selectively and in combination with other control methods in order to secure their effectiveness for the long term.

Fire is one of the most effective tools for restoring temperate grasslands that are dominated by weeds. Historically, grasslands are ecosystems that are accustomed to frequent fire events, and altered fire regimes in Australia (Stuwe & Parson, 1977), New Zealand (Mark, 2007; Standish *et al* ., 2009), the United States (Foster & Gross 1998; Stromberg 2007), and South Africa (Sankaran & Anderson, 2009) have been linked to the modification of these landscapes (Archer *et al* ., 1988; Knicker, 2007). Fire quickly creates available space for heat resistant seeds to germinate and grow relatively free of competition (Meyer & Schiffman, 1999), and a number of studies have observed that fire significantly reduces weed species and promotes native recruitment (Huddleson & Young, 2005; Prober *et al* ., 2005; Bryant *et al* ., 2017). Lipoma *et al* . (2018) identified fire to significantly reduce the viable number of seeds in the soil compared to pre-burnt conditions, and as most weeds often have dense seedbanks, this can be beneficial in reducing at least the surface seedbanks of some species (Peltzer & Douglass, 2019). In contrast, some species, particularly broadleaf weeds such as *Echium plantagineum* , are promoted by fire (Prober *et al* ., 2004). Heat tolerance in seeds has been linked to seed shape, with more rounded seeds demonstrating higher resistance than thinner seeds in European temperate grasslands (Ruprecht *et al* ., 2015). This suggests that follow up weed management of burnt sites is critical for the successful establishment of native species. In Australia, a summer wildfire was observed to kill 90% of the standing native spear-grass (Austrastipa spp.), which is considered relatively fire tolerant (Sinclair *et* *al* ., 2014). Fire also offers soil manipulation services as carbon and nitrogen volatize at 180 and 200^{O}C respectively (DiTomaso *et al* ., 2006), therefore hot fires can remove soil nutrients that advantage annual weeds and further inhibit their re-establishment (Knicker, 2007). Strategically burning when problematic weeds are actively growing can effectively prevent seed set for that season (Prober *et al* ., 2005). The complexity of fire effects suggests that post management plans should be specific for the site in order to promote the establishment of a healthy native grassland community (Musil *et al* ., 2005; DiTomaso *et al* ., 2006).

Soil manipulation

Altered soil nutrients and textures resulting from agriculture have important consequences on the ability for standing vegetation to take up water and nutrients (Sankaran & Anderson, 2009), therefore restoring these factors to resemble historic levels can be important for weed suppression. Soil nutrients such as nitrogen, phosphorous and potassium, are altered by agricultural practises, and even after agriculture has ceased, the soil nutrient levels remain higher than historical levels (Prober *et al* ., 2005). Annual weeds become problematic in environments with high nitrogen, where they are able to quickly dominate over the slower-growing native perennial grasses (Huddleson & Young, 2005). Perennials invest in developing deeper rots systems that allow them to store and recycle nutrients, giving established perennials an advantage over annuals in areas of low nutrient availability. Therefore, integrating control methods that target soil nutrient levels should be strongly considered for those grassland restoration projects in areas that have a history of agriculture. This can be achieved with the addition of a carbon source, such as sucrose, can increases soil microbial activity reduces soil nutrients, and leaves them unavailable for used by nutrient-adapted weeds. This technique has been used successfully in Australia (Prober *et al* ., 2005; Hacker *et al* ., 2011) and the United States (Blumenthal *et al* ., 2003). In one reported prairie restoration, carbon addition reduced soil nitrogen by 86%, which subsequently reduced weed biomass by 54% (Blumenthal*et al* ., 2003). While carbon addition has proven to be successful, it is a time and resource-demanding approach. Prober *et al* . (2004) used 500g of sugar for every square metre, which was reapplied every three months, making this technique difficult to implement at a landscape scale. Further, it is only suitable with nitrophillic weeds (Blumenthal *et al* ., 2003).

Another method for altering soil dynamics is through mechanical disturbance techniques, such as tilling or scalping. These techniques are effective for creating an environment that promotes the establishment of broadcasted seeds and reduces competition from weeds (Tikka *et al* ., 2001). As many weed seeds respond positively to disturbance events, tillage can be used to stimulate stored seedbanks (Stromberg *et al* ., 2007). Scalping is a technique where top soil is removed from a site and subsequently treated. This is a useful technique in highly degraded sites that are heavily infested by weeds as it removes their seedbank as well as the elevated nutrient levels that promote their growth and establishment (Brown *et al* ., 2017). Consequently, scalping may result in excessive waste soil, increases erosion rates, habitat loss and disrupted mycorrhizal symbiosis, and therefore should be implemented with caution (Gibson-Roy *et al* ., 2010; Gerlach, 2015; Brown *et al* ., 2017). Further, weed reinvasion can occur on scalped sites, and Gerlach (2015) identified weeds to occupy 70% of the ground cover after three years of scalping and revegetation. Scalping treatments followed by with spot-spraying has proven to be successful within small scale (1m X1m plots) for reducing all vegetation (Gibson-Roy *et al* ., 2010).

Revegetation

Establishing dense competition from desirable species is the most effective way to reduce weeds and return natural ecosystem functionality. In the case that natural regeneration is an unviable option, competition can be introduced using a variety of methods including direct seeding (Thomas *et al* ., 2019; Cole *et al* ., 2005), transfer of threshing material (Baasch *et al* ., 2016) or hay (Sengel *et al* ., 2016), direct drilling (Bakker *et al* ., 2003) and plant plugs (Tikka *et al* ., 2001). Hedberg & Kotowski (2010) reviewed the effectiveness of different revegetation options for fragmented grasslands and found that direct seeding (sowing and broadcasting) to be the most widely used and most effective for introducing species back to semi-natural systems. However, they specifically recommended the use of plant plugs for the establishment of rarer species (Hedberg & Kotowski, 2010).

The effectiveness of species richness in seed mixes has been explored for grassland restorations, with both high and low rates demonstrating beneficial results dependent on the projects scope (Prober & Thiele, 2005; Wortley *et al* ., 2013). The determined species mix is often reflective of the goals of that particular restoration project; for example, Conrad & Tishew (2011) found a high seed mix of 35 species achieved their goals of increasing species diversity as well as establishing target species, whilst in another area, Huddleson & Young (2005) used a mix of only three native grasses to successfully outcompete weeds. Further, it appears that high species diversity improves the establishment of native species (Barr *et al* ., 2017), long-term resilience to weed reinvasion (Carter & Blair, 2012; Scotton, 2016), and provide habitat for recolonization of threatened wildlife (McDougal & Morgan, 2005). Nemec *et al* . (2013) has demonstrated seed diversity to be a more important factor than seed rate for achieving reasonable competition for weeds. While high seed rates can improve the chances in successfully outcompeting weeds (Tikka *et al* ., 2001; Bakker *et al* ., 2003; Barr *et al* ., 2017), this approach can waste seeds as a result of higher intraspecific competition, and the associated high costs can make it unpractical (Sheley *et al* ., 2006; Wagner *et al* ., 2011). Seed mixes low in diversity and density can promote spontaneous secondary succession, and this can stimulate ecosystem processes more quickly (Lengyel*et al* ., 2012). We note that the failure of sown seeds to establish can be linked to several factors, including herbivory, adverse weather conditions, and species competition (Gibson-Roy *et al* ., 2007), therefore implementing pre-sowing management that minimises these threats is critical. Whilst it is clear that the introduction of seeds or seedlings is often critical for the restoration of many degraded temperate grasslands it is also clear that the best implementation method will be dependent on the site, scale and funding available to the project (Prober & Thiele, 2005).

Grazing management

Grazers play an important role in the continuous removal of leaf litter and generating available space for new recruitment (Lengyel *et al* ., 2012; Török *et al* ., 2018), which can promote species richness (Towne *et al* ., 2005; Klaus *et al* ., 2018). Germination of the native North American prairie grass, *Nassella pulchra* was enhanced by burning and sheep grazing (Dyer, 2002). Moderate grazing (30-50 within a 303ha enclosure) from *Bos bison*significantly improved the species richness of a Prairie grassland within its later stages of development (approximately 10-years after revegetation) (Wilsey & Martin, 2015), this was also observed within tallgrass prairies (Towne *et al* ., 2005). Livestock can transport seeds of important species over great distances via endo or ectozoochory if remnant sites are available (Lengyel *et al* ., 2012; Török*et al* ., 2018). Further, the careful management of paddock rotations for grazing livestock has been identified to be critical in maintaining genetic diversity for plants threatened by fragmentation (Plue *et al* ., 2019). High genetic diversity can allow for a population to react more responsively to disturbance events through improved resilience.

Overgrazing by livestock is one of the leading causes of grassland degradation (Bartolome *et al* ., 2009; Zha & Gao, 2011; Wortley*et al* ., 2013). The effects of different degrees of overgrazing were observed by Török *et al* . (2018) within four different Hungarian steppe grassland communities. They found that the highest richness was achieved from low grazing (less than one animal per hectare), but medium was also suitable (1-2.5 animals per hectare), and grazing densities above this had detrimental effects of species richness. Further, the different grassland communities responded differently to the grazing intensities suggesting they are grassland specific (Török *et al* ., 2018). Competition dynamics between forb and grass species were altered by livestock grazing in southern Argentina (Díaz Barradas *et al* ., 2001). Under sheep grazing, the grasses did not produce inflorescences and forbs became taller and more abundant compared to non-grazing tracts, where grass species dominated (Díaz Barradas *et al* ., 2001). Forb cover was also observed to increase in Prairie grasslands when exposed to grazing from cattle and bison (Towne *et al* ., 2005). Therefore, grazing intensities should be carefully managed, particularly during drought periods to promote competition from native perennial grasses (Klaus *et al* ., 2018), and resting paddocks from grazing when natives are emerging, particularly if herbage is sparse, could improve their establishment and survival (Clarke & Davison, 2014).

Considerations for long-term management

The role of grassland seedbanks

Limitations in funding and technology makes it difficult to manage grassland restoration projects over long time scales (Freudenberger & Gibson-Roy, 2011; Lengyel *et al* ., 2012). Without follow-up management, weeds can re-establish either from regeneration from the seedbank or from migration of seeds from surrounding sites (Gibson-Roy*et al* ., 2007). By knowing how long the dominant weeds seeds remain viable within the soil seedbank, we can make recommendations as to how long a site should be actively managed.

Weed species are often prolific seed producers, therefore, to protect a rehabilitated grassland from being reinvaded by a dominant weed species, an understanding of its seedbank persistence is required. Gardener*et al* ., (2003) identified that after three years of no seed migration within a *Nassella neesiana* dominated grassland, there was still, on average, 1457 viable seeds per square metre of this species, indicating that ongoing management of only three years would not be effective for controlling this invasive grass. Seed longevity studies are useful for investigating how long a species can persist at different depth or soil types. The common trend is that seed viability declines with shallow burial and increased duration within the seedbank. This is seen in *Andropogon gayanus* (Bebawi *et al* ., 2018),*R. raphanistrum* (Reeves *et al* ., 1981), *Conyza canadensis* (Vargas *et al* ., 2018) and *Artemisia tridentate*(Wijayratne & Pyke, 2012). Seeds located on or just below the soil surface are exposed to more intense fluctuations in soil moisture and temperature compared to deeper buried seeds, and these fluctuations can result in the shallow buried seeds drying out. A two-year study found*A . tridentate* had only 0-11% of seeds remaining viable at the soil surface compared with almost half of the seeds maintaining viability at 3cm depth (Wijayratne & Pyke, 2012). Seed predation by mammals, birds, or soil microbes is also enhanced at these shallow depths (Dalling *et al* ., 2011). Seeds that are buried deeper into the soil profile, are often better protected from these devitalizing and predatory pressures (Bebawi *et al* ., 2018). The trade-off, however, is that at these depths seed dormancy is usually prolonged, particularly for photoblastic seeds (Benvenuti *et al.* , 2005; Ahmed *et al.* , 2015), making these weeds troublesome for managing cropping systems utilizing tillage, since this can resurface viable invasive seeds, resulting in reinvasion.

Anticipating climate Change

Grasslands currently store approximately 34% of the worlds terrestrial carbon, making these ecosystems important carbon sinks and play a critical role in climate change mitigation (Contant, 2010). Carbon sequestration is achieved by grassland vegetation holding organic carbon within their roots, therefore higher sequestration is found in less disturbed grasslands with long lived perennial grasses that develop dense root systems (Acharya *et al* ., 2012). This suggests that long term management of grasslands will likely provide greater climate regulation. Carbon sequestration has been observed to improve with good management techniques, particularly the addition of nitrogen fixing plants (De Deyn *et al* ., 2011), addition of fertilizers and lime (Acharya *et al* ., 2012) and withholding excessive grazing (Eze*et al* ., 2018). Further, grasslands are essential for human food security and provide an income for approximately 1.3 billion people around the world (Suttie *et al* ., 2005). Livestock grazing utilizes 80% of the total agricultural land and contributes to 40% of global agricultural production (Suttie *et al* ., 2005). It is predicted that demands for animal-based proteins and dairy are only going to increase as a result of projected population growth (O'Mara, 2012), making functioning grasslands critical for providing adequate nutritional resources.

Accelerated climate change adds a further element of complexity for managing restoration projects into the future. It is expected that for every 1^{O}C increase in air temperature, there will be a 1.5^{O}C increase in soil temperature (Ooi *et al* ., 2011), which may also cause disruptions to the seedbanks of many plant species. Temperature has proven to be an important environmental factor for breaking seed dormancy and these increased temperatures could influence these important physiological processes (Ooi, 2012; Prosotto*et al* ., 2014). Further, atmospheric CO_2 has steadily risen from 325ppm recorded in 1970 to 405ppm in 2017 (Lindsay, 2018), and this is expected to approximately double by the end of this century (IPCC, 2019). Enhanced atmospheric CO_2 can result in higher saturation of CO_2, potentially reducing photorespiration in C_3 plants, even under a warmer climate. This increased physiological efficiency has been demonstrated to alter dynamics between C_3 and C_4plants (Dukes, 2000). As a result of these physiological improvements, such as increased water-use efficiency (Varga *et al* ., 2015), plants can allocate more resources to growth and fecundity and these changes have also been observed to be more pronounce in weeds than natives or crops (*Marble et al* ., 2015). Changes in extreme weather patterns is expected to increase as a result of human induced climate change. Compared to pre-industrial data, changes in the intensity and pattern of rainfall events are already being noticed (Power *et al* ., 2017). Changes in rainfall have direct consequences on the intensity and frequency of fire, drought and flood events (Ooi, 2012). As these factors play an important role in shaping the vegetation of ecosystems and agroecosystems, new challenges for managing native and weed competition dynamics can be expected.

Conclusion

Temperate grasslands are now significantly degraded throughout the world as a result of human actions. Weeds now dominate many of these degraded systems and act to hold them in this undesirable altered state. A number of successful restoration techniques have been developed to reduce weed dominance and promote native species, but it is clear that a single technique for restoration is rarely successful for the long term. In order to reduce dominant weeds, we must continue to research the integration of control methods that are economical, practical and applicable to temperate grasslands at a local, regional and global scales. Researchers should also aim to develop long-term studies that observe successional changes in plant dynamics as a result of various treatments. It is critical that managers plan now for changes in weather patterns (such as rainfall frequency and intensity) as a result accelerated climate change. This review recognises the similarities in successful temperate grassland restorations involve the ongoing effort of targeting the above and below ground density of the dominant weeds. Revegetation methods are often site- and study-specific and depend on proximity of remanent vegetation, budget and restoration goals.

Conflict of Interest

The Authors declare no conflict of interest.

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