

CROSS-BORDER COLONISTS

How naturalised species reshape prairies,
Pampas, and savannas

*Wayne Dawson, Dana Blumenthal, Jan Čuda, Richard P. Duncan,
Danielle Frohlich, Martin Hejda, Yanjie Liu, Ana Novoa,
Ayub M.O. Oduor, Jan Pergl, Pavel Pipek, Vânia R Pivello,
Petr Pyšek, David M. Richardson, Kateřina Štajerová,
SynHab contributors (Paulina Anastasiu, Michael Ansong,
Margarita Arianoutsou, Julie F. Barcelona, Suneeta Bhatta,
Farzaneh Bordbar, Israel Borokini, Josef Brůna,
Laura Celesti-Gradow, Eduardo Chacón-Madrigal, Dorjee,
Franz Essl, Estrela Figueiredo, Guillaume Fried, Nicol Fuentes,
Christian Gilli, Michael Glaser, José Ramón Grande Allende,
Zigmantas Gudžinskas, Rachael Holmes, Philip E. Hulme, Inderjit,
Eun Su Kang, Daniel W. Krix, Ingolf Kühn, Trobjon Makhkamov,
Desika Maříková-Moodley, Alfred Maroyi, Rachid Meddour,
Pierre Meerts, Sharif A. Mukul, Brad R. Murray, Megan L. Murray,
Daniel Nickrent, Prince E. Norman, Ali Omer, Annette Patzelt,
Pieter B. Pelsler, Joan Pino, Marc Riera, Julissa Rojas-Sandoval,
Roser Rotchés-Ribalta, Stepan Senator, Bharat Babu Shrestha,
Gideon F. Smith, Sima Sohrabi, Barbara Tokarska-Guzik,
Montserrat Vilà, Viktoria Wagner, Ayşe Yazlık, Elena Zykova),
and Alessandra Körtz*

Introduction

Invasive species are typically defined as species introduced by humans (intentionally or unintentionally) to a region outside their historical native range, that have subsequently established self-sustaining populations (naturalised) and spread widely, negatively impacting biodiversity and ecosystem impacts [1]. Biological invasions are a major component of global environmental change [2]. Every continent harbours invasive species representing many taxa, and new invasions will likely increase along with global trade volumes and connectivity [3].

Human-made or natural disturbances can promote biological invasions through competitor removal and increased resource availability [4]. While cold or arid climates can explain the persistence of primary grassland (see Chapter 3), elsewhere they are maintained by regular disturbance, through fire, wild or domestic animal herbivory or trampling (see Chapters 13 and 14). Grasslands, savannas, and rangelands (referred to collectively as ‘grasslands’) are also subject to human-wrought disturbances, including fertilisation and soil ploughing. Natural or human disturbances combined with high propagule pressure (the number of individuals introduced and frequency of introduction events) increase the probability of introduced species becoming naturalised and invasive [5]. High propagule pressure could result from direct seeding with forage species, or propagules dispersing from nearby sites. Although well documented by scientific literature for many parts of the world, non-native plant invasions have not been summarised and reviewed across grasslands worldwide.

Here, we first provide a summary of data on naturalised and invasive plants found in grassland habitats, available from the SynHab database [6], which catalogues introduced plant species occurrences in different habitat types from multiple sources at the country or state/province level (referred to as ‘regions’) based on information in published floras. Although habitat information from floras is not standardised, our approach provides a first approximate estimation of naturalised plant species numbers and identities in grasslands. We then review documented examples of invasive plants and their impacts in grassland ecosystems for each continent. Our plant focus reflects the volume of plant invasion research [7], but we also dedicate a section to animal invasions. Finally, we consider how grassland invaders are managed. We use the term ‘grassland’ broadly, covering primary and secondary grasslands that may or may not be grazed by livestock, savannas, and rangelands (see Chapter 3).

Naturalised and invasive plants in the SynHab database

We used naturalised plant species records in grasslands that represented 84 regions. Species were assigned the status of ‘invasive’ for a region if described as such by the data source, and we grouped species within ‘continents’ according to the World Geographic Scheme for Recording Plant Distributions (Africa, Antarctic, Asia [Temperate+Tropical], Europe, Northern America, Southern America, Pacific Islands) [8]. Life form (annual/biennial versus perennial) was obtained from the GloNAF database [9] and the World Checklist of Vascular Plants [10,11], which we also used to standardise species names and families. We specifically answered four questions with this dataset.

What are the most widely naturalised plant species in grassland habitats?

We identified 11 species occurring in 23 to 17 regions (Table 16.1). *Erigeron canadensis* – a wind-dispersed weed of disturbed areas – is the most widespread. All but one species (*Lantana camara* agg.) are herbaceous, with six from temperate Eurasia. *Cynodon dactylon* is widely introduced in the Americas as a turf grass, and *Paspalum dilatatum* was introduced to many regions for forage and grazing.

How many naturalised and invasive plant species are there per continent?

SynHab records 2,722 naturalised plant species in grasslands. Except for the Antarctic and Pacific, all other continental regions have more than 500 naturalised plant species in grasslands, and of these only Northern America has fewer than 100 plant species that are considered invasive (Table 16.2). Note that the percent coverage of continents by regions with data in the database is variable, which

Table 16.1 The 11 most widely naturalised plant species in grassland habitats, according to the SynHab database [6]

# Regions	Species	Family	Life form	Origin
23	<i>Erigeron canadensis</i>	Asteraceae	Annual/biennial	North America
19	<i>Trifolium repens</i>	Fabaceae	Perennial	Eurasia
19	<i>Plantago lanceolata</i>	Plantaginaceae	Perennial	Eurasia
19	<i>Taraxacum officinale</i> agg.	Asteraceae	Perennial	Eurasia
18	<i>Cirsium vulgare</i>	Asteraceae	Annual/biennial	Eurasia/Africa
18	<i>Cynodon dactylon</i>	Poaceae	Perennial	Eurasia/Australia/ Africa
18	<i>Poa annua</i>	Poaceae	Annual/biennial	Eurasia
17	<i>Lantana camara</i> agg.	Verbenaceae	Perennial (shrub)	Tropical America
17	<i>Oxalis corniculata</i>	Oxalidaceae	Perennial	Asia
17	<i>Lolium perenne</i>	Poaceae	Annual/biennial	Eurasia/Africa
17	<i>Paspalum dilatatum</i>	Poaceae	Perennial	South America

Regions = number of regions around the world where a species is known to be naturalised in grasslands.

Table 16.2 Number of naturalised and invasive plant species known to occur in grasslands of regions with data available from the SynHab database, per continent

Continent (% area represented by regions)	Grassland formations	#Naturalised species; # invasive species
Australasia (13)	ASFMG, MSGFM, TGMS, TLSGS	548; 131
Africa (41)	MSGFM, TGMS, TLSGS, TMSGs, WSDSG	933; 139
Europe (34)	ASFMG, CSDSG, TGMS	820; 150
Northern America (23)	CSDSG, MSGFM, TGMS, WSDSG	514; 51
Southern America (19)	CSDSG, TGMS, TLSGS, TMSGs	635; 114
Asia (49)	ASFMG, CSDSG, TGMS, TMSGs	417; 189
Antarctic (14)	–	88; 0
Pacific (9)	–	9; 0

Also shown are the % of each continent's area (ice-free for mainland Antarctica) represented by SynHab regions, and the International Vegetation Classification natural grassland formations represented (see Chapter 3): ASFMG = Alpine Scrub, Forb Meadow & Grassland; CSDSG = Cool Semi-Desert Scrub & Grassland; MSGFM = Mediterranean Scrub, Grassland & Forb Meadow; TGMS = Temperate Grassland, Meadow & Shrubland; TLSGS = Tropical Lowland Shrubland, Grassland & Savanna; TMSGs = Tropical Montane Shrubland, Grassland & Savanna. Secondary grasslands are likely present everywhere. Note: Antarctica is represented only by the Falklands/Malvinas.

will affect absolute numbers, and species records will include both secondary and primary grasslands. Estimated numbers of naturalised and invasive plants will change as more habitat data become available from floras (especially for Southern America, the Pacific, and Australasia).

What is the representation of annuals/biennial species compared to perennials?

Only 6% of vascular plant species globally are annual [12], and we assess if this proportion is reflected among naturalised and invasive plants of grasslands. Most naturalised plant species were perennials for six continents, while Asia, Australasia, and Southern America had slim majorities for annual/biennial species (Figure 16.1a). Invasive annuals/biennials formed the majority

How naturalised species reshape prairies, Pampas, and savannas

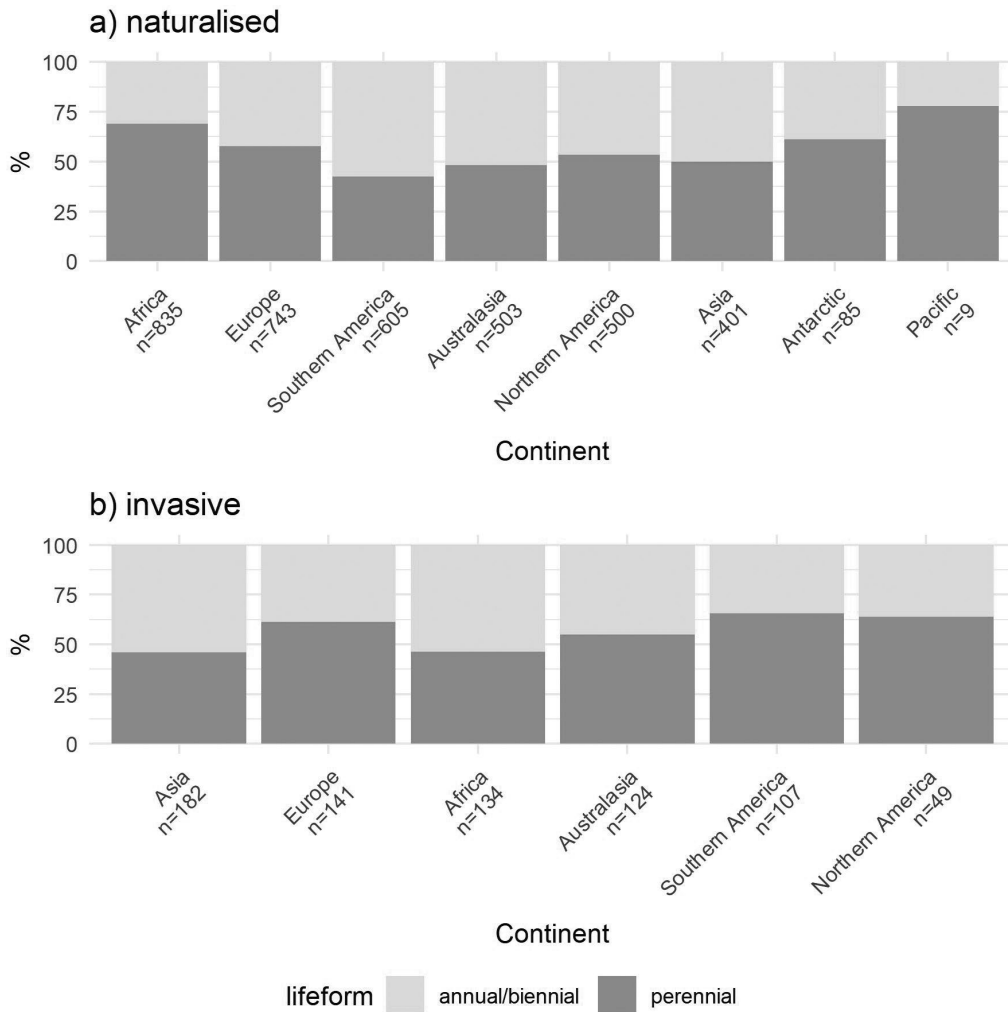


Figure 16.1 Proportion of (a) naturalised and (b) invasive plant species in grasslands, which are annual/biennial or perennial, per continent. Life form information was available for 2,620 species naturalised in grasslands somewhere in the world. *n* = number of species per continent.

in only Asia and Africa (Figure 16.1b). Only 22% of all known naturalised plants worldwide are annual [13], so percentages of annual/biennial species found in grasslands are relatively high. This could reflect high propagule pressure of (largely unintentionally) introduced annual agricultural weeds, and an ability of annuals to capitalise on grasslands' frequent disturbance events and pulses of increased resource availability [13].

Which plant families are the most represented by naturalised plant species per continent?

Naturalised Poaceae reached the top-ten most represented families in Southern and Northern America, Europe, Temperate Asia, and the Antarctic, but the Brassicaceae (mostly herbaceous),

Iridaceae (geophytes), and Rosaceae (woody and non-woody perennials) were better represented than Poaceae in most continents (Figure 16.2). Many Brassicaceae species were probably introduced as crop or agricultural weed species to new continents, while many Iridaceae and Rosaceae species were introduced as garden ornamentals.

In summary, based on the data available, the most globally widespread naturalised plant species of grasslands tend to be herbaceous, likely due to intentional or unintentional introductions into secondary grasslands for agriculture [14]. While the majority of species in grasslands are perennial for most continents, annual/biennial species representation is high compared to global floras. Comparing life histories and growth forms of naturalised plants with native grassland floras might

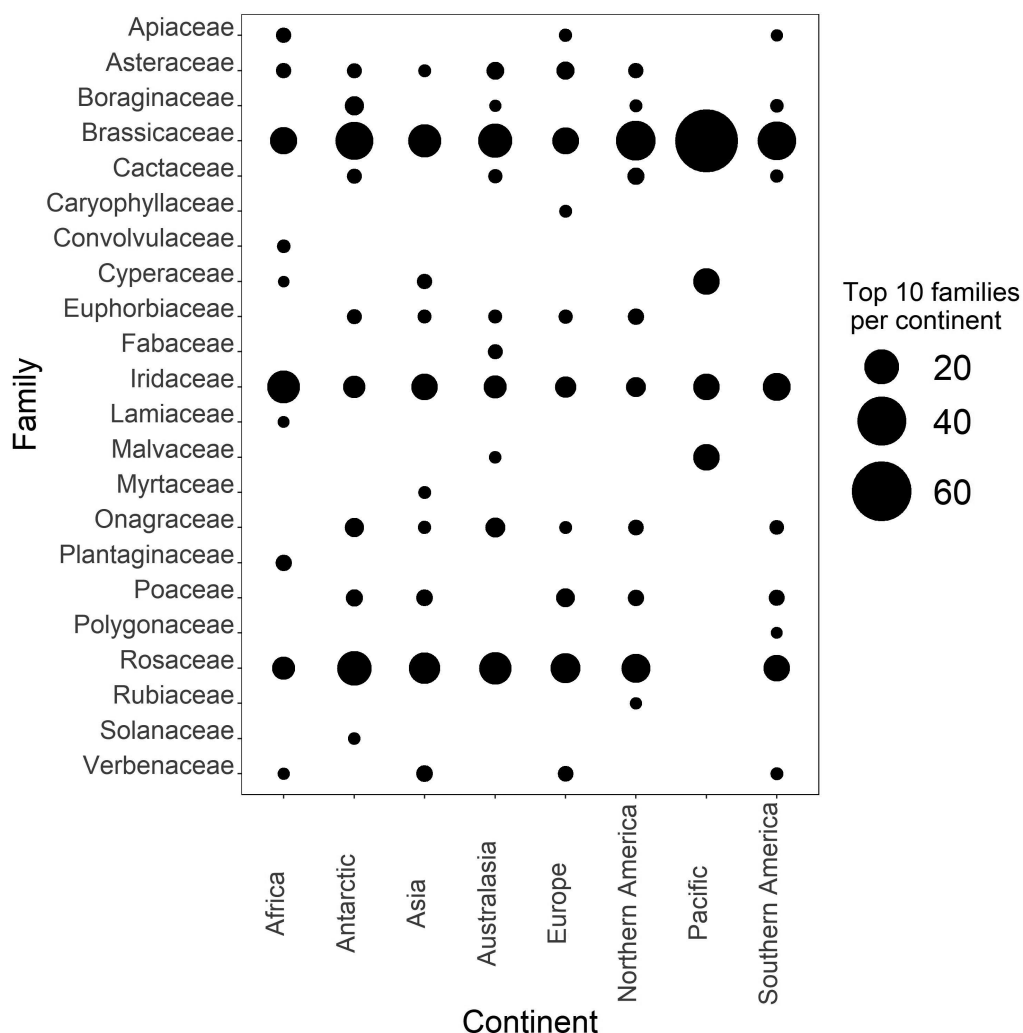


Figure 16.2 Top-ten most represented plant families with naturalised species in grasslands per continent. Circles represent the percentage of all naturalised species in the grasslands of a continent from each family.

reveal if these attributes play a role in grassland invasion success. Finally, the Poaceae and Asteraceae are represented by relatively low numbers of species recorded in grasslands. However, species numbers cannot capture the impacts that individual species can have on invaded grasslands. Thus, we review documented examples of impactful invasive plants (per continent) and then animals, in grassland systems.

Examples of invasive species in grasslands per continent

Africa

In Africa, most research on invasive plants comes from Southern and East Africa. In South Africa, invaders of the ‘Grassland Biome’ [15] comprise mainly woody species, though many occur primarily in riparian habitats within the biome, where their success is driven by river dynamics [16]. South African grasslands also have a notable paucity of grass taxa in important invader lists [17]. Several shrub species have spread very rapidly over large parts of South Africa’s grassland biome, the best example being pompom weed (*Campuloclinium macrocephalum*) which spread very rapidly throughout most parts of the biome since it was first reported in the 1960s [18]. *Pyracantha angustifolia* has rapidly invaded and transformed especially high-altitude grasslands [19]. Woody plant invasions in South African grasslands have had major hydrological impacts, due to their deep roots and the massive increases in biomass they generate [20]. Such impacts justify large-scale invasive plant management through the national Working for Water programme.

In East Africa, the annual herb *Parthenium hysterophorus* disrupts native plant pollination networks [21], and reduces native plant diversity in Tanzania’s rangelands [22], Ugandan savanna [23], and Ethiopian grazing lands [24]. However, native plant species may evolve and eventually coexist with *P. hysterophorus* [25]. The cactus *Opuntia stricta* is widespread and abundant in parts of Kenya, Tanzania, and Ethiopia [26]. In Nairobi National Park, *L. camara* invasion alters soil chemical properties [27], while *Opuntia ficus-indica* can facilitate native plants through protection from intense ungulate herbivory [28]. In Kenya and Ethiopia, the drought-resistant tree *Neltuma juliflora* (Mesquite) has encroached on arid and semi-arid pastureland, reducing native plant species diversity [29,30].

Plant invasions can impact people’s livelihoods in Africa. *O. stricta* in Kenya [26], the perennial herb *Chromolaena odorata* in Tanzania [27], and *L. camara* in Uganda [27] all threaten local livelihoods through effects on livestock grazing and forage quality. In Kenya [29,31] and Ethiopia [30], *N. juliflora* invasion is associated with several negative socioeconomic impacts in pastoral communities (income loss, human and livestock poisoning, snakebite risk, and allergic reactions), but can also confer multiple benefits as materials, charcoal, and soil restoration [30,31].

Antarctic

Ice-free locations on the Antarctic peninsula, and nearby and more distant sub-Antarctic islands harbour several invasive plant species, notably *Poa annua* [32]. In South Georgia, *P. annua* and several other naturalised European grasses and forbs are colonising and spreading into areas recently deglaciated due to warming [33]. This region may develop novel communities with northern hemisphere grass species as an important component. South Georgia has native grass-dominated vegetation on its coastal fringes and peninsulas, and it is unknown if these species-poor communities will become more vulnerable to invasion as the climate warms. In the Falklands/Malvinas,

extensive grasslands are under threat from invasion by the evergreen, bird-dispersed spiny shrub *Berberis darwinii* (Calafate), native to southern Chile and Argentina [34].

Asia

Asia is home to vast grassland systems (see Chapter 3), and most invasion research comes from China and India, with knowledge gaps for temperate grasslands of the former Soviet Union. The Steppe of Mongolia and Northern China harbours at least 108 introduced plant species [35,36], and there are at least two impactful invasive plants: the North American grass *Cenchrus longispinus* and the perennial *Solanum rostratum*. *C. longispinus* has sharply barbed fruits that can injure grazing livestock, and its spines degrade fur and wool product quality. Solanine (a neurotoxin) in *S. rostratum* fruits renders them fatal to livestock when ingested, and the plant threatens biodiversity in China's northeast grasslands, outcompeting native plants under high nitrogen deposition [37]. *Hordeum jubatum* and *Oenothera biennis* are naturalising in disturbed steppe and saline grasslands of temperate Asia [38]. Large areas of Indian savanna are vulnerable to multiple invasive plant species [39], including *Neltuma laevigata*, *Mikania micrantha* and *C. odorata* [39]. In Nepal, *Mikania* is invading tall and short grasslands in Chitwan National Park [40], while the shrub *L. camara* invades protected areas of Sri Lankan grassland [41]. *P. hysterophorus* has been a problematic weed of South Asian grasslands and rangelands [42] since its accidental introduction as a cereal grain contaminant. The forb *Sida rhombifolia* is invading Northern Iranian grasslands, and is spread by livestock and hay movement, but is itself unpalatable [43]. Montane grasslands also harbour invasive plants, e.g., the perennial herb *Centaurea iberica* in Himalayan grasslands [44].

Several invasive plants in Asian grassland systems impact wild mammal herbivores or livestock forage and grazing quality. For example, in India's Banni grassland, *N. laevigata* inhibits native forage grass growth [45], though native grasses have recovered after tree removal. In Nepal, *M. micrantha* smothers and outcompetes native food plants of the greater one-horned rhinoceros (*Rhinoceros unicornis*) [40]. Invasive *Lantana* may be replacing grasslands in Pandangaran nature reserve (Java), with implications for fire regimes and wild buffalo (*Bos javanicus*) habitat suitability [46].

Australasia

While human disturbance and conversion of vegetation occurred in Australia and New Zealand long before British colonialism [47,48], it was after Europeans' arrival and expansion of pastoralism in the 1800s that invasive grasses were widely introduced to increase pastoral productivity [49,50]. At least 330 and 250 grass species have been recorded as naturalised in Australia and New Zealand, respectively. Many native grasslands managed for grazing contain a high proportion of impactful invasive grasses that are often among the most problematic invaders. Invasive grasses often outcompete and reduce the diversity of native plants, particularly at more productive sites. Some grasses can reduce forage quality, for example, *Nassella trichotoma* is a perennial tussock-forming grass invasive in both Australia and New Zealand, which is unpalatable to stock and forms extensive monocultures. Invasive grasses can also modify the frequency and intensity of fire, and thus act as ecosystem transformers. In this regard, two species stand out as particularly problematic in Australia: *Cenchrus ciliaris* (Buffel grass) and *Andropogon gayanus* (Gamba grass).

C. ciliaris dominates many semi-arid pastoral lands with the potential to invade two-thirds of mainland Australia [51]. This grass forms dense swards, outcompeting native species, and

altering fire frequency and intensity by increasing fuel loads and creating greater connectivity in otherwise sparsely vegetated areas [52]. *C. ciliaris* is, however, economically important to the livestock industry and is not listed in the weed legislation of any Australian state, with only a few regional-level attempts to manage its spread and impacts [51]. Since large-scale plantings began in northern Australia in the 1980s, *A. gayanus* has spread rapidly in savanna ecosystems. The natural fire regime in this region involved frequent low-intensity ground fires, which maintained a grass-savanna woodland. *A. gayanus*, however, grows up to 4 m tall and produces up to five times the biomass of native grass species, massively increasing the fuel load [53]. The resulting increase in fire intensity can kill adult woody plants and substantially alter community structure [54].

Europe

Data from vegetation plots across Europe indicate low to intermediate levels of invasion compared to other habitats and continents [55,56]. Even the widespread and locally abundant invasive herbaceous plants *Heracleum mantegazzianum* [57], *Solidago* species [58], *Senecio* species, and *Reynoutria japonica* [59] occur relatively infrequently at the European scale [56]. Many European native plants have been moved within the continent. The most frequently occurring neophyte (introduced after 1500) in grassland plots was the southeastern European herbaceous forage legume *Onobrychis viciifolia* (common sainfoin), closely followed by the tall-statured, wind-dispersed North American forbs *Erigeron annuus* and *E. canadensis* [56]. *E. annuus* and several other herbaceous dicot species are invading meadows and open grasslands of the northern Steppe in European Russia [60]. However, across Europe, no species listed above occurred in more than 1.5% of grassland plots [56]. Only one tree species (*Robinia pseudoacacia*) was among the top 20 most frequently occurring species in grasslands in European regions [56]. Other notable examples of woody invaders include *Prunus serotina* in ‘rough’ grassland, and *Rosa rugosa* in sand dune communities [59]. From the habitat perspective, grasslands with sandy soils are more frequently invaded (up to >30% of vegetation plots) than other grassland types, maybe because the unstable substrates lead to disturbance gaps, aiding invader colonisation [56]. Further work is needed to understand why other European grasslands are less invaded.

North America

The desert steppe in the U.S. and Mexico, and Pacific coast grasslands have been subjected to novel grazing and fire regimes, and millions of hectares are dominated by invasive annual grasses [61–63]. *Bromus tectorum* (cheatgrass) is one of several short-lived, fire-adapted grasses that have altered fire regimes [64,65], leading to positive feedbacks and further invasion [66]. Dominance by *B. tectorum* is associated with two- to four-fold higher fire frequency and larger fires [63,64]. Impacts also include reduced plant and animal diversity, forage availability for livestock and wildlife, and carbon storage [66–68]. Other invasive annual grasses include *Avena barbata*, *Bromus hordeaceus*, and *Hordeum murinum* in California grasslands, and *Bromus rubens* and *Taeniatherum caput-medusae* in the southwestern U.S. In contrast, large areas of semi-arid prairie of the western Great Plains and western montane grasslands still exist as diverse native grasslands with lower levels of invasion [69].

Invasive perennial grasses, forbs, and trees also have substantial impacts in North American grasslands. Many species, such as *Agropyron cristatum*, *Bromus inermis*, and *Poa pratensis*, have been planted for livestock forage or soil stabilisation and can form dense monocultures, reducing plant diversity [69,70]. In northern Mexico, *C. ciliaris* spreads unassisted from plantings,

increasing fire frequency, reducing plant and animal diversity, productivity, and carbon storage [71]. North American grasslands can become dominated by perennial forbs such as *Euphorbia esula* and *Centaurea stoebe* in the Great Plains and foothill grasslands, and *Centaurea solstitialis* in California grasslands [72]. Grassland conversion to shrublands or forests is also common in some regions. For example, a single tree species, *Triadica sebifera*, is largely responsible for converting much of the endangered southeastern U.S. coastal prairie to forest [73].

Pacific Islands

Pacific island grasslands are novel communities, largely composed of invasive introduced grass species. Transformation of lowland areas from mosaics of woody species-dominated vegetation types to grasslands is directly attributable to human arrival and subsequent conversion of lowland plant communities to agricultural systems. The Hawaiian Islands provide the best-studied example of vegetation conversion to invader-dominated grassland in the Pacific, and this development arose historically after Polynesian arrival and European colonisation.

While Polynesian settlers played a pivotal role in reshaping the landscape of Pacific islands by using fire to clear areas for agriculture and housing [74,75], arrival of Europeans greatly accelerated the conversion to grasslands, and water resource diversion, to accommodate sugarcane cultivation. Scrubland was repurposed as grazing terrain for cattle and other grazing animals. Deliberate or accidental introduction of fire-adapted grasses (e.g., *Cenchrus setaceus*, *C. ciliaris*, *Megathyrsus maximus*), which burn swiftly and recover rapidly following fire, prevented native vegetation recovery. This process initiates a grass/fire cycle wherein grasses replace native woody species, degrading biodiversity in invaded areas [66].

South America

South American grassland ecosystems include tropical and temperate grasslands and open savannas (see Chapter 3), and some are severely affected by intentionally introduced invasive plants that are facilitated by human disturbance (e.g., ranching and silviculture). Most information available on grassland invasions in South America comes from non-Andean grassland ecosystems.

In tropical savannas and grasslands, multiple African C4 grasses were introduced to improve pastures for livestock feeding thanks to their exceptional productivity and nutritional value [76,77]. *Melinis minutiflora*, *Hyparrhenia rufa*, *M. maximus*, *A. gayanus*, and several *Urochloa* species have become major invaders in the savanna-like ecosystems of the Brazilian Cerrado and Colombian-Venezuelan Llanos. These species often competitively exclude native plants, reduce biodiversity [76], alter nutrient cycling and carbon dynamics [78], and greatly increase fire risk due to high dry biomass production [79]. In temperate grasslands – the Pampas of southern Brazil, Uruguay, and northeast Argentina – *Eragrostis plana* and *C. dactylon* reduce native herb diversity, while *E. plana* is of low forage quality [80,81]. *Arundo donax* – a giant grass of likely Asian origin – is highly dispersed in South American grazing lands [82]. Under climate change, several grasses are expected to spread further [83].

Pinus trees were brought to South America mostly for timber and pulp production or, in some cases, as ornamentals [84], and include North and Central American [76,84,85], and Mediterranean species [84,85]. The invasion success of most species lies in their high reproductive capacity and multiple introduction events [84,86]. Pine invasions in grasslands and savannas change vegetation structure, altering light and water availability to native plants, nutrient cycling, and fire regimes [87].

Invasive animals

While literature on invasive animals and their impacts in grasslands is more limited than for plants, documented examples of invasive animals fall into four key groups: mammal herbivores, omnivores and carnivores, and insects (especially ants).

A comprehensive dataset on invasive mammals in the Neotropics [88] records significant mammal invasions in grasslands, particularly in Argentina, Brazil, Chile, Paraguay, and Uruguay, including the herbivores European hare (*Lepus europaeus*) and feral horses (*Equus ferus*). Feral horses are also controversially considered invasive in North America, having been extinct for ~10,000 years prior to reintroduction in the sixteenth century, and have substantial impacts on plant communities, wildlife, and soils [89]. On Pacific islands, goats and sheep (*Capra aegagrus hircus* and *Ovis* spp.) and deer species, including Asian spotted deer (*Axis axis*) overgraze and trample vegetation, compact soils, opportunistically consume native ground-nesting bird eggs, and disperse invasive plant seeds [90,91]. Native grasslands in Australia and New Zealand are also vulnerable to the impacts of invasive mammalian herbivores. Horses, goats, and several deer species have significant impacts on Australasian grasslands through grazing and trampling (e.g., in the Australian Alps [92]). In South America, *A. axis* was introduced to Argentina in the early 1900s for hunting, became invasive, and is advancing in Brazil [93]. For Europe, despite comprehensive reports and listing, there is scant information available on the impacts of invasive mammals in grasslands compared to other continents: many of the 16 invasive mammal species in the EU List of Union Concern are not typically grassland species [94].

European rabbit (*Oryctolagus cuniculus*) impacts have been considerable in multiple countries. Rabbits were introduced to Australia and New Zealand in the 1800s and contributed to major grassland degradation. In central Chilean matorral, rabbits graze more on native forbs in open grassland areas, resulting in an altered spatial distribution of forb populations under shrub canopies [95]. Native grasslands on the sub-Antarctic Macquarie Island were overgrazed by rabbits, leading to exposure of bare ground and erosion of soil and peat [96]. A rabbit eradication programme has been successful, with no sightings since 2011 [97].

Wild pigs (*Sus scrofa*) are omnivores of conservation concern in South America, with potential to establish widely in Cerrado and Pampas [98]. Wild pigs are also increasingly problematic in New Zealand, the U.S. and Mexican grasslands [99]. Their impacts include competition with and depredation of native wildlife species, disease transmission to livestock and wildlife, and increases in disturbance with corresponding effects on native vegetation and plant invasion [99–101]. Similar impacts are reported from the Pacific [91], where novel grasslands also harbour rats (*Rattus* spp.) and mice (*Mus* spp.), which prey on native bird eggs and chicks [102], disperse introduced plant seeds, and consume the seeds of native species [103].

Examples of invasive mammal carnivores are less common than for herbivores. Feral dogs (*Canis familiaris*) can have impacts on native mammals. Indian foxes (*Vulpes bengalensis*) in dry grasslands of India suffer interference competition from feral dogs, with the result that foxes avoid more disturbed human-dominated grassland areas despite higher rodent prey availability [104]. Feral dogs are also listed as impacting South American grassland systems through competition with and predation on native animals [88]. In New Zealand, the early introduction of mustelids (weasels, stoats, ferrets) to combat rabbits has been disastrous for native animals.

Several studies report surprising impacts of invasive ants, including trophic cascade impacts of *Pheidole megacephala* on a Kenyan savanna [105], which disrupted native ant-*Vachellia* mutualisms. Consequently, *Vachellia* tree browsing by elephants (*Loxodonta africana*) increased, resulting in reduced tree density and increasing visibility for plains zebra (*Equus quagga*), ultimately

leading to lions (*Panthera leo*) switching to other prey species. A succession of introduced ant species has invaded grassland systems in the southeastern U.S. and Texas [106,107]. In Texan savanna, invasive red imported fire ants (*Solenopsis invicta*) displaced native fire ants, disrupted spatial patterns, and reduced diversity of native ant assemblages in wet, undisturbed environments [108]. The more recently introduced crazy ant (*Nylanderia fulva*) can displace *Solenopsis*, reducing native arthropod species richness and abundance in Texan coastal prairie [106]. A notable invasive insect in several Asian countries is the fall armyworm (*Spodoptera frugiperda*) [109]. While primarily an agricultural pest, the armyworm is highly polyphagous (with >350 food plants) and can occur in grasslands. Further research is needed to shed light on the extent and impact of insect invasions in grassland ecosystems [107].

Management of grassland invaders

Prevention of invasive species introduction to a new area is always the most cost-effective approach to preventing invasions. Effective legislation and biosecurity measures can minimise introduction risk [110], but preventing all introductions of potentially invasive species will be impossible. Thus, risk management of invasions in grasslands should prioritise prevention of invasive species incursions and removal of nascent populations in areas of high conservation and socioeconomic value [111]. Legislation can also codify and enforce species eradication and control. For example, *N. trichotoma* and *N. neesiana* are classed as weeds of national significance in Australia [112], a federal designation that requires a strategic weed management plan coordinated across government. Eradication of invasive animals is more realistic on islands and for vertebrates (e.g., rabbits on Macquarie Island). For prioritised plant species, there are typically four sets of tools for eradication or control: mechanical, chemical (herbicides), biological (including biological control and grazing), and fire.

Mechanical control of invasive grasses has been tried with relative success, such as mowing and clipping in South America [113], and mowing in Türkiye to prevent seed production of spiny species [6]. In New Zealand, legislation in the 1940s established a government-funded programme aimed at controlling the grass *N. trichotoma* by hand-grubbing on properties [114]. For smaller tree populations of species that do not resprout after cutting or from roots, uprooting young plants and cutting the mature ones close to the ground can be effective, although frequent fires have been applied to kill pine trees in Brazil [115].

Chemical control of invasive grasses in South America with herbicides has been tried alone [116] or with other techniques such as fire and uprooting [117]. In North America, invasive annual grass management includes landscape-scale herbicide treatment. In Australia, *A. gayanus* was listed as a weed of national significance in 2012, and under legislation in several states must be eradicated or managed where it occurs, primarily through herbicide treatment.

Grazing has been attempted for multiple invasive plant species across continents. In Brazilian pastures, management through continuous or rotational grazing might reduce the invasion of *E. plana* [80]. Targeted grazing and restoration of native communities have been used to biologically control invasive annual grasses in semi-arid regions of North America. In South African grasslands, grazing exclusion sometimes promotes invasions, but overall, human changes to fire regimes and overgrazing seem to be more important than grazing cessation [118].

Classical biological control is a 'last-resort' management tool for invasive species that are too widespread and abundant for eradication to be possible. A cochineal bug (*Dactylopius opuntiae* 'stricta' biotype, Hemiptera) was introduced to Kenya in 2014 to control *O. stricta* [119], which has shown significantly reduced reproduction at local scales. In East Africa, the parthenium beetle *Zygogramma bicolorata* shows promise as a potentially effective control agent of

P. hysterophorus [120,121]. Invasive animals have also been subjected to biological control: strains of calicivirus were introduced from the 1980s to Australia and New Zealand, replacing the earlier myxoma virus as a successful rabbit control agent.

The Hawaiian Islands provide examples of approaches to managing invasive grasses and associated fire risk [122]. Restoration efforts can use native species with fire-resistant characteristics [123], and prescribed fire can be employed to control grasses, especially in areas where native shrubs and grasses once co-occurred [124]. Experimental grazing may reduce fuel loads and subsequent fire intensity effectively [125]. Whatever the region and management tool, the often vast areas invaded, and the effects of other global change drivers (climate change, nutrient deposition) represent major challenges to eradication and spread prevention.

Conclusions

Based on available data, widespread naturalised species in grasslands tend to be herbaceous. All continents harbour naturalised plant species in grassland, many are annual/biennial, and several families are well represented consistently across continents. However, only a small subset of naturalised plants will be impactful invaders in primary grassland systems: further work is needed to identify invaders of primary versus secondary grasslands, and whether they differ in the attributes that make them successful. Examples of invasive plants in grasslands range from trees and shrubs to tall-statured, fast-growing, and disturbance-adapted annual or perennial herbs. Invasive grasses feature heavily on most continents, but not in Africa. Invasive plant impacts include changes to vegetation structure (especially trees and shrubs) and fire regimes (especially the grasses), reduced native plant diversity and forage availability for both wild mammal herbivores and domestic livestock, in what are often multiple-use systems. Invasive mammal herbivores, pigs and ants, can have pervasive impacts in grasslands, and should be priority targets for management. Some invasive taxa have similar impacts in multiple continents, for example, the plants *P. hysterophorus* and *Neluma* spp., African grasses (especially *C. ciliaris*), the European rabbit and multiple deer species, and feral dogs and pigs. Sharing experiences across borders should serve to promote best practices in the management of these impactful invaders.

References

1. Convention on Biological Diversity. 2023. Invasive Alien Species, Glossary of Terms. <https://www.cbd.int/invasive/terms.shtml>
2. IPBES. 2023. Summary for policymakers of the thematic assessment report on invasive alien species and their control of the intergovernmental science-policy platform on biodiversity and ecosystem services. In: Roy HE, Pauchard A, Stoett P, Renard Truong T, Bacher S, Galil BS, Hulme PE, Ikeda T, Sankaran KV, McGeoch MA, et al., editors. *Thematic assessment report on invasive alien species and their control of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (IPBES). Bonn, Germany: IPBES secretariat, <https://doi.org/105281/zenodo7430692>
3. Seebens H, Blackburn TM, Dyer EE, Genovesi P, Hulme PE, Jeschke JM, Pagad S, Pyšek P, Winter M, Arianoutsou M, et al. 2017. *Nat Commun.* 8:14435. <https://doi.org/10.1038/ncomms14435>
4. Davis MA, Grime JP, Thompson K. 2000. *J Ecol.* 88:528–534. <https://doi.org/10.1046/j.1365-2745.2000.00473.x>
5. Lockwood JL, Cassey P, Blackburn T. 2005. *Trends Ecol Evol.* 20:223–228. <https://doi.org/10.1016/j.tree.2005.02.004>
6. SynHab. Macroecology of plant invasions: global synthesis across habitats. 2023. SynHab. <https://www.synhab.com>
7. Pyšek P, Richardson DM, Pergl J, Jarošík V, Sixtová Z, Weber E. 2008. *Trends Ecol Evol.* 23:237–244. <https://doi.org/10.1016/j.tree.2008.02.002>

8. Brummitt RK. 2001. *World geographical scheme for recording plant distributions*, 2nd ed. Pittsburgh, PA: Hunt Institute for Botanical Documentation.
9. van Kleunen M, Pyšek P, Dawson W, Essl F, Kreft H, Pergl J, Weigelt P, Stein A, Dullinger S, König C, et al. 2019. *Ecology*. 100:e02542. <https://doi.org/10.1002/ecs.2542>
10. Brown MJM, Walker BE, Black N, Govaerts RHA, Ondo I, Turner R, Nic Lughadha E. 2023. *New Phytol*. 240:1355–1365. <https://doi.org/10.1111/nph.18919>
11. Govaerts R, Nic Lughadha E, Black N, Turner R, Paton A. 2021. *Sci Data*. 8:215. <https://doi.org/10.1038/s41597-021-00997-6>
12. Poppenwimer T, Mayrose I, DeMalach N. 2023. *Nature*. 624:109–114. <https://doi.org/10.1038/s41586-023-06644-x>
13. Pyšek P, Pergl J, Essl F, Lenzner B, Dawson W, Kreft H, Weigelt P, Winter M, Kartesz J, Nishino M, et al. 2017. *Preslia*. 89:203–274. <https://doi.org/10.23855/preslia.2017.203>
14. Firm J, Moore JL, MacDougall AS, Borer ET, Seabloom EW, HilleRisLambers J, Harpole WS, Cleland EE, Brown CS, Knops JMH, et al. 2011. *Ecol Lett*. 14:274–281. <https://doi.org/10.1111/j.1461-0248.2010.01584.x>
15. Mucina L, Rutherford MC, editors. 2006. *The vegetation of South Africa, Lesotho and Swaziland*. Pretoria: South African National Biodiversity Institute.
16. Rouget M, Hui C, Renteria J, Richardson DM, Wilson JR. 2015. *S Afr J Bot*. 101:24–31. <https://doi.org/10.1016/j.sajb.2015.04.009>
17. Visser V, Wilson JR, Fish L, Brown C, Cook GD, Richardson DM. 2016. *Glob Ecol Biogeogr*. 25:679–692. <https://doi.org/10.1111/geb.12445>
18. Goodall J, Witkowski ETF, Morris CD, Henderson L. 2011. *Biol Invasions*. 13:2217–2231. <https://doi.org/10.1007/s10530-011-0035-6>
19. Adams LD, Giovannoni D, Clark VR, Steenhuisen S-L, Martin GD. 2023. *Plants*. 12:1308. <https://doi.org/10.3390/plants12061308>
20. Le Maitre DC, Blignaut JN, Clulow A, Dzkiti S, Everso CS, Gorgens AHM, Gush MB. Impacts of plant invasions on terrestrial water flows in South Africa. In: van Wilgen B, Measey J, Richardson D, Wilson J, Zengeya T, editors. *Biological invasions in South Africa. Invading nature - Springer series in invasion ecology*, vol 14. Cham: Springer. pp. 431–457. https://doi.org/10.1007/978-3-030-32394-3_15
21. Ojija F, Arnold SEJ, Treydte AC. 2019. *Arthropod Plant Interact*. 13:719–734. <https://doi.org/10.1007/s11829-019-09701-3>
22. Musese LJ, Andrew SM, Shirima DD, Witt A, Kilewa R. 2020. *Int J Eng Technol Manag Res*. 7:11–20. <https://doi.org/10.29121/ijetmr.v7.i5.2020.631>
23. Nuwagira C, Tumusiime J, Kagoro GR. 2020. *Afr J Plant Sci*. 14:402–410. <https://doi.org/10.5897/AJPS2020.2043>
24. Boja M, Girma Z, Dalle G. 2022. *Diversity*. 14:675. <https://doi.org/10.3390/d14080675>
25. Oduor AMO. 2022. *New Phytol*. 233:983–994. <https://doi.org/10.1111/nph.17574>
26. Shackleton RT, Witt ABR, Piroris FM, Van Wilgen BW. 2017. *Biol Invasions*. 19:2427–2441. <https://doi.org/10.1007/s10530-017-1453-x>
27. Shackleton, RT., Witt, ABR., Aool, W., Prat, CF. 2017. *Afr J Range Forage Sci*. 34:1–11. <https://doi.org/10.2989/10220119.2017.1301551>
28. Oduor AMO, Long H, Fandohan AB, Liu J, Yu X. 2018. *Biol Invasions*. 20:2745–2751. <https://doi.org/10.1007/s10530-018-1757-5>
29. Maundu P, Kibet S, Morimoto Y, Imbumi M, Adeka R. 2009. *Biodiversity*. 10:33–50. <https://doi.org/10.1080/14888386.2009.9712842>
30. Ilukor J, Rettberg S, Treydte A, Birner R. 2016. *Pastoralism*. 6:14. <https://doi.org/10.1186/s13570-016-0061-1>
31. Mwangi E, Swallow B. *Invasion of Prosopis juliflora and local livelihoods: case study from the lake Baringo area of Kenya*. Nairobi: World Agroforestry Centre.
32. Leihy RI, Peake L, Clarke DA, Chown SL, McGeoch MA. 2023. *Sci Data*. 10:200. <https://doi.org/10.1038/s41597-023-02113-2>
33. Tichit P, Brickle P, Newton RJ, Convey P, Dawson W. 2024. *NeoBiota*. 92:85–110. <https://doi.org/10.3897/neobiota.92.117226>
34. UKOTs Team. 2023. Brahms Online. <https://brahmsonline.kew.org/falklands>
35. Baasanmunkh S, Urgamal M, Oyuntsetseg B, Sukhorukov AP, Tsegmed Z, Son DC, Erst A, Oyundelger K, Kechaykin AA, Norris J, et al. 2022. *PhytoKeys*. 192:63–169. <https://doi.org/10.3897/phytokeys.192.79702>

36. Ma JS, Li HR. 2018. *The checklist of alien invasive plants in China*. Beijing, China: Higher Education Press.
37. Sun J-K, Liu M-C, Tang K-Q, Tang E-X, Cong J-M, Lu X-R, Liu Z-X, Feng Y-L. 2023. *Front Plant Sci*. 14:1169317. <https://doi.org/10.3389/fpls.2023.1169317>
38. Abramova LM. 2012. *Russ J Ecol*. 43:352–357. <https://doi.org/10.1134/S1067413612050037>
39. Mungi NA, Qureshi Q, Jhala YV. 2023. *J Appl Ecol*. 60:2400–2412. <https://doi.org/10.1111/1365-2664.14506>
40. Murphy ST, Subedi N, Jnawali SR, Lamichhane BR, Upadhyay GP, Kock R, Amin R. 2013. *Oryx*. 47:361–368. <https://doi.org/10.1017/S003060531200124X>
41. Sampson C, Leimgruber P, Tonkyn D, Pastorini J, Janaka HK, Sotherden E, Fernando P. 2018. *Biol Conserv*. 220:50–59. <https://doi.org/10.1016/j.biocon.2018.01.021>
42. Bhatta S, Shrestha BB, Pyšek P. 2023. *NeoBiota*. 88:135–167. <https://doi.org/10.3897/neobiota.88.104118>
43. Sohrabi S, Gherekloo JZ, Nezamabadi N. 2023. *Iran Nature*. 8:81–90. <https://doi.org/10.22092/irn.2023.360646.1487>
44. Reshi Z, Rashid I, Khuroo AA, Wafai BA. 2008. *Trop Ecol*. 49:147–156.
45. Singh YD, Kumar VV, Patel IL, Sharma YK, Dangar SV, Subba Rao SV. 1998. *Ecorestoration of Banni grassland*. India: Gujarat Institute of Desert Ecology.
46. Peh KS-H. 2010. *Biodivers Conserv*. 19:1083–1099. <https://doi.org/10.1007/s10531-009-9755-7>
47. Fletcher M-S, Hall T, Alexandra AN. 2021. *Ambio*. 50:138–149. <https://doi.org/10.1007/s13280-020-01339-3>
48. McWethey DB, Wilmschurts JM, Whitlock C, Wood JR, McGlone MS. 2014. *PLoS One*. 9:e111328. <https://doi.org/10.1371/journal.pone.0111328>
49. van Klinken RD, Panetta FD, Coutts S, Simon BK. 2015. *Biol Invasions*. 17:565–579. <https://doi.org/10.1007/s10530-014-0749-3>
50. Lonsdale WM. 1994. *Austral J Ecol*. 19:345–354. <https://doi.org/10.1111/j.1442-9993.1994.tb00498.x>
51. Grice AC, Friedel MH, Marshall NA, Van Klinken RD. 2012. *Environ Manage*. 49:285–294. <https://doi.org/10.1007/s00267-011-9781-6>
52. Clarke PJ, Latz PK, Albrecht DE. 2005. *J Veg Sci*. 16:237–248. <https://doi.org/10.1111/j.1654-1103.2005.tb02361.x>
53. Setterfield SA, Rossiter-Rachor NA, Hutley LB, Douglas MM, Williams RJ. 2010. *Divers Distrib*. 16:854–861. <https://doi.org/10.1111/j.1472-4642.2010.00688.x>
54. Setterfield SA, Clifton PJ, Hutley LB, Rossiter-Rachor NA, Douglas MM. 2018. *Sci Reports*. 8:6628. <https://doi.org/10.1038/s41598-018-24704-5>
55. Pyšek P, Chytrý M, Jarošík V. 2010. Habitats and land use as determinants of plant invasions in the temperate zone of Europe. In: *Bioinvasions and globalization: ecology, economics, management, and policy*. Oxford: Oxford University Press. p. 66–79. <https://doi.org/10.1093/acprof:oso/9780199560158.003.0006>
56. Axmanová I, Kalusová V, Danihelka J, Dengler J, Pergl J, Pyšek P, Večeřa M, Attorre F, Biurrun I, Boch S, et al. 2021. *J Veg Sci*. 32:e12994. <https://doi.org/10.1111/jvs.12994>
57. Pyšek P, Pyšek A. 1995. *J Veg Sci*. 6:711–718. <https://doi.org/10.2307/3236442>
58. Szymura M, Świercz S, Szymura TH. 2022. *Land Degrad Dev*. 33:1985–1998. <https://doi.org/10.1002/ldr.4278>
59. Dassonville N, Vanderhoeven S, Vanparys V, Hayez M, Gruber W, Meerts P. 2008. *Oecologia*. 157:131–140. <https://doi.org/10.1007/s00442-008-1054-6>
60. Sukhorukov AP. 2011. *Feddes Repert*. 3–4:287–304. <https://doi.org/10.1002/fedr.201100004>
61. Mack RN, Thompson JN. 1982. *Am Nat*. 119:757–773. <https://doi.org/10.1086/283953>
62. Seabloom EW, Harpole WS, Reichman OJ, Tilman D. 2003. *Proc Natl Acad Sci U S A*. 100:13384–13389. <https://doi.org/10.1073/pnas.1835728100>
63. Bradley BA, Curtis CA, Fusco EJ, Abatzoglou JT, Balch JK, Dadashi S, Tuanmu M-N. 2018. *Biol Invasions*. 20:1493–1506. <https://doi.org/10.1007/s10530-017-1641-8>
64. Balch JK, Bradley BA, D’Antonio CM, Gómez-Dans J. 2013. *Glob Chang Biol*. 19:173–183. <https://doi.org/10.1111/gcb.12046>
65. Fusco EJ, Finn JT, Balch JK, Nagy RC, Bradley BA. 2019. *Proc Natl Acad Sci U S A*. 116:23594–23599. <https://doi.org/10.1073/pnas.1908253116>
66. D’Antonio CM, Vitousek PM. 1992. *Ann Rev Ecol Evol Syst*. 23:63–87. <https://doi.org/10.1146/annurev.es.23.110192.000431>

67. Nagy RC, Fusco EJ, Balch JK, Finn JT, Mahood A, Allen JM, Bradley BA. 2021. *J Appl Ecol.* 58: 327–337. <https://doi.org/10.1111/1365-2664.13770>
68. Rice PM. 2005. Downy brome, *Bromus tectorum* L. In: Celestine A, Duncan, Janet K, Clarke editors. *Invasive plants of range and wildlands and their environmental, economic, and societal impacts*. Lawrence, KS: Weed Science Society of America. p. 147–170.
69. Zapisocki Z, de Assis Murillo R, Wagner V. 2022. *Rangel Ecol Manag.* 83:20–30. <https://doi.org/10.1016/j.rama.2022.02.011>.
70. Vaness BM, Wilson SD. 2007. *Can J Plant Sci.* 87:1023–1028. <https://doi.org/10.4141/CJPS07120>.
71. Burquez A, Martinez-Yrzar A, Miller M, Rojas K, Quintana M de los A, Yetman D. 1998. Mexican grasslands and the changing aridlands of Mexico: an overview and a case study in northwestern Mexico. In: Tellman B, Finch DM, Edminster C, Hame R, editors. *The future of arid grasslands: identifying issues seeking solutions*. Rocky Mountain Research Station Fort Collins, Colorado: USDA Forest Service. p. 21–32.
72. Lejeune KD, Seastedt TR. 2001. *Conserv Biol.* 15:1568–1574. <https://doi.org/10.1046/j.1523-1739.2001.00242.x>
73. Grace JB, Smith MD, Grace SL, Collins SL, Stohlgren TJ. 2001. Interactions between fire and invasive plants in temperate grasslands of North America. In: Galley KEM, Wilson TP, editors. *Proceedings of the invasive species workshop: the role of fire in the control and spread of invasive species*. Tallahassee, FL: Tall Timbers Research station. p. 40–65.
74. Kirch PV, Ellison J. 1994. *Antiquity.* 68:310–321. <https://doi.org/10.1017/S0003598X00046615>
75. Athens JS. 2004. *Rec Aust Mus.*, Supplement. 29:15–30. <https://doi.org/10.3853/j.0812-7387.29.2004.1398>
76. Pivello VR, Shida CN, Meirelles ST. 1999. *Biodiv Conserv.* 8:1281–1294. <https://doi.org/10.1023/A:1008933305857>
77. Zenni RD, Herrera I, De Sá Dechoum M, Ziller SR, De Matos ACL, Núñez CI, Núñez MA, Pauchard A. 2022. Plant Invasions in South America. In: Clements DR, Upadhyaya MK, Joshi S, Shrestha A, editors. *Global plant invasions*. Cham: Springer. p. 187–208. https://doi.org/10.1007/978-3-030-89684-3_9
78. Garcia DB, Xavier RO, Camargo PB, Vieira SA, Pivello VR. 2022. *Flora.* 286:151968. <https://doi.org/10.1016/j.flora.2021.151968>
79. Gorgone-Barbosa E, Pivello VR, Bautista S, Zupo T, Rissi MN, Fidelis A. 2015. *Biol Invasions.* 17:423–431. <https://doi.org/10.1007/s10530-014-0740-z>
80. Baggio R, Medeiros RBD, Focht T, Boavista LDR, Pillar VD, Müller SC. 2018. *Perspect Ecol Conserv.* 16:158–165. <https://doi.org/10.1016/j.pecon.2018.06.004>
81. Garcia S, Guido A, Pezzani F, Lattanzi FA. 2023. *Austral Ecol.* 48:1107–1120. <https://doi.org/10.1111/aec.13341>
82. Goolsby JA, Moran PJ, Martínez Jiménez M, Yang C, Canavan K, Paynter Q, Ota N, Kriticós DJ. 2023. *Invasive Plant Sci Manag.* 16:81–109. <https://doi.org/10.1017/inp.2023.17>
83. Lopes A, Demarchi LO, Piedade MTF, Schöngart J, Wittmann F, Munhoz CBR, Ferreira CS, Franco AC. 2023. *Perspect Ecol Conserv.* 21:128–135. <https://doi.org/10.1016/j.pecon.2023.02.005>
84. Simberloff D, Núñez MA, Ledgard NJ, Pauchard A, Richardson DM, Sarasola M, Van Wilgen BW, Zalba SM, Zenni RD, Bustamante R, et al. 2010. *Austral Ecol.* 35:489–504. <https://doi.org/10.1111/j.1442-9993.2009.02058.x>
85. Richardson DM, Williams PA, Hobbs RJ. 1994. *J Biogeogr.* 21:511. <https://doi.org/10.2307/2845655>
86. Miashike RL, Kortz AR, Zarate Do Couto HT, Pivello VR. 2021. *Austral Ecol.* 46:722–736. <https://doi.org/10.1111/aec.12987>
87. De Abreu RCR, Durigan G. 2011. *Plant Ecol Divers.* 4:269–278. <https://doi.org/10.1080/17550874.2011.594101>
88. Rosa CAD, Ribeiro BR, Bejarano V, Puertas FH, Bocchiglieri A, Barbosa ALDS, Chiarello AG, Paglia AP, Pereira AA, Moreira AFDS, et al. 2020. *Ecology.* 101:e03115. <https://doi.org/10.1002/ecy.3115>
89. Eldridge DJ, Ding J, Travers SK. 2020. *Biol Conserv.* 241:108367. <https://doi.org/10.1016/j.biocon.2019.108367>
90. Hess SC, Judge SW. 2021. *Pacific Sci.* 75:561–573. <https://doi.org/10.2984/75.4.8>
91. Hess SC. 2016. *Mammal Study.* 41:47–60. <https://doi.org/10.3106/041.041.0202>
92. Hartley R, Blanchard W, Schroder M, Lindenmayer DB, Sato C, Scheele BC. 2022. *Conserv Sci Practice.* 4:e601. <https://doi.org/10.1111/csp2.601>
93. Preuss JF, Posser E, Albrecht LB, Da Silva VPR, Bandiera FC. 2020. *CheckList.* 16:1139–1142. <https://doi.org/10.15560/16.5.1139>

94. Tedeschi L, Biancolini D, Capinha C, Rondinini C, Essl F. 2022. *Mammal Review*. 52:252–266. <https://doi.org/10.1111/mam.12277>
95. Jaksic FM, Fuentes ER. 1980. *J Ecol*. 68:665. <https://doi.org/10.2307/2259427>
96. Costin AB, Moore DM. 1960. *J Ecol*. 48:729. <https://doi.org/10.2307/2257346>
97. Australian Government. 2014. *Macquarie Island: from rabbits and rodents to recovery and renewal (Fact Sheet)*. Department of the Environment.
98. La Sala LF, Burgos JM, Caruso NC, Bagnato CE, Ballari SA, Guadagnin DL, Kindel A, Etges M, Merino ML, Marcos A, et al. 2023. *J Nat Conserv*. 73:126393. <https://doi.org/10.1016/j.jnc.2023.126393>
99. Shwiff S, Pelham A, Shwiff S, Haden-Chomphosy W, Brown VR, Ernst K, Anderson A. 2020. *Biol Invasions*. 22:3101–3117. <https://doi.org/10.1007/s10530-020-02311-8>
100. Tolleson DR, Pinchak WE, Rollins D, Hunt LJ. 1995. *Feral hogs in the rolling plains of Texas: perspectives, problems, and potential*. Great Plains Wildlife Damage Control Workshop Proceedings. Oklahoma: Noble Foundation. p. 124–128. <https://digitalcommons.unl.edu/gpwcwcp/454>
101. Tierney TA, Cushman JHall. 2006. *Biol Invasions*. 8:1073–1089. <https://doi.org/10.1007/s10530-005-6829-7>
102. Work TM, Duhr M, Flint B. 2021. *J Wildl Dis*. 57: 125–131. <https://doi.org/10.7589/JWD-D-20-00065>
103. Shiels AB, Drake DR. 2015. *Ann Bot PLANTS*. 7:plv057. <https://doi.org/10.1093/aobpla/plv057>
104. Vanak AT, Gompper ME. 2010. *J Appl Ecol*. 47:1225–1232. <https://doi.org/10.1111/j.1365-2664.2010.01870.x>
105. Kamaru DN, Palmer TM, Riginos C, Ford AT, Belnap J, Chira RM, Githaiga JM, Gituku BC, Hays BR, Kavwele CM, et al. 2024. *Science*. 383:433–438. <https://doi.org/10.1126/science.adg1464>
106. LeBrun EG, Abbott J, Gilbert LE. 2013. *Biol Invasions*. 15:2429–2442. <https://doi.org/10.1007/s10530-013-0463-6>
107. Wills BD, Landis DA. 2018. *Oecologia*. 186:323–338. <https://doi.org/10.1007/s00442-017-4007-0>
108. LeBrun EG, Plowes RM, Gilbert LE. 2012. *J Anim Ecol*. 81:884–895. <https://doi.org/10.1111/j.1365-2656.2012.01954.x>
109. Mendesil E, Tefera T, Blanco CA, Paula-Moraes SV, Huang F, Viteri DM, Hutchison WD. 2023. *J Plant Dis Prot*. 130:1175–1206. <https://doi.org/10.1007/s41348-023-00777-x>
110. Reaser JK, Burgiel SW, Kirkey J, Brantley KA, Veatch SD, Burgos-Rodríguez J. 2020. *Biol Invasions*. 22:1–19. <https://doi.org/10.1007/s10530-019-02156-w>
111. Booy O, Mill AC, Roy HE, Hiley A, Moore N, Robertson P, Baker S, Brazier M, Bue M, Bullock R, et al. 2017. *Biol Invasions*. 19:2401–2417. <https://doi.org/10.1007/s10530-017-1451-z>
112. Humphries T, Florentine SK. 2021. *Plants*. 10:1036. <https://doi.org/10.3390/plants10061036>
113. Gorgone-Barbosa, E. 2009. Eficiência do manejo no controle de duas espécies de gramíneas invasoras em cerrados paulistas. [MSc Dissertation]. São Paulo: Universidade de São Paulo. <https://teses.usp.br/teses/disponiveis/41/41134/tde-21092009-135818/pt-br.php>
114. Lamoureaux S, Bourdôt G. 1999. Nassella tussock management in New Zealand. *12th Australian Weed Conference*. West Point Convention Centre, Hobart, Tasmania, Australia. p. 16–19. <https://caws.org.nz/old-site/awc/1999/awc199910161.pdf>
115. Pauchard A, García R, Zalba S, Sarasola M, Zenni R, Ziller S, Nuñez MA. 2015. 14. Pine invasions in South America: reducing their ecological impacts through active management. In: Canning-Clode J, editor. *Biological invasions in changing ecosystems*. Poland: De Gruyter Open. p. 318–342. <https://doi.org/10.1515/9783110438666-020>
116. Durigan G, Contieri WA, Franco GADC, Garrido MAO. 1998. *Acta Bot Bras*. 12:421–429. <https://doi.org/10.1590/S0102-33061998000400011>
117. Martins CR, Hay JDV, Scaléa M, Malaquias JV. 2017. *Acta Bot Bras*. 31:546–554. <https://doi.org/10.1590/0102-33062016abb0433>
118. O'Connor TG, van Wilgen BW. 2020. The impact of invasive alien plants on rangelands in South Africa. In: van Wilgen B, Measey J, Richardson D, Wilson J, Zengeya T, editors. *Biological invasions in South Africa. Invading Nature - Springer series in invasion ecology*, vol 14. Cham: Springer. pp. 459–487. https://doi.org/10.1007/978-3-030-32394-3_16
119. Witt ABR, Nunda W, Makale F, Reynolds K. 2020. *BioControl*. 65:515–523. <https://doi.org/10.1007/s10526-020-10018-x>
120. Kanagwa W, Kilewa R, Treydte AC. 2020. *Biocontrol Sci Technol*. 30:806–817. <https://doi.org/10.1080/09583157.2020.1768219>
121. Mersie W, Alemayehu L, Strathie L, McConnachie AST, Negeri M, Zewdie K. 2019. *Biocontrol Sci Technol*. 29:239–251. <https://doi.org/10.1080/09583157.2018.1545220>

122. Frohlich, DS. 2009. *Effects of Pennisetum setaceum control methods on plant community development and invasion by Senecio madagascariensis in a degraded Hawaiian dry forest*. University of Hawaii at Manoa.
123. Loh R, Ainsworth A, Tunison T, D'Antonio C. 2009. *Testing native species response to fire at Hawai'i Volcanoes National Park*. Honolulu, HI: University of Hawai'i.
124. Castillo JM, McAdams A. 2006. *Effects of prescribed grazing and burning treatments on fire regimes in alien grass-dominated wildland-urban interface areas, Leeward Hawaii*. Kamuela, HI: US Fish and Wildlife Service.
125. Blackmore M, Vitousek PM. 2000. *Biotropica*. 32:625–632. <https://doi.org/10.1111/j.1744-7429.2000.tb00509.x>