Golden Sun Moth (*Synemon plana*)

Conservation Assessment Project

Literature Review prepared on behalf of Trust for Nature, August 2022

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Report:	Golden Sun Moth (*Synemon plana*) Conservation Assessment Project		
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Cover photo:

Golden Sun Moths (*Synemon plana*). Photo by Lucy Gibson (2022).

Executive Summary

Synemon plana is a diurnal Castniid moth endemic to southeast Australia, which requires highly specific weather and climactic conditions for emergence and reproduction during the summer (Kutt *et al.*, 2015). Since its EPBC Listing in 2002 *S. plana* has remained poorly known and accordingly its populations in Victoria are in need of an integrated synthesis of working knowledge in order to assign priorities to its ongoing conservation management. The presence of abundant *S. plana* populations within sheep grazing land have raised questions regarding the response of the species to traditional habitat restoration approaches within these landscapes.

Victoria's *S. plana* populations are largely represented by poorly documented rural populations and fragmented urban populations, with both categories facing ongoing threats from development (Biosis 2019b). More than 98% of Victoria's native grassy ecosystems have been radically altered by anthropogenic activities since early European settlement (DSEWPC 2011). *S. plana* has specific biomass requirements that can be easily upset by inappropriate land use practices (ACT Government 2020a).

The suitability of *S. plana's* habitat depends on a raft of intersecting biotic and abiotic factors (Kutt *et al.*, 2015). Natural *S. plana* habitat is characterised by the presence of native C3 grasses, especially Wallaby Grasses (*Rytidosperma* spp.) (Kutt *et al.*, 2016). Many such remnant landscapes have historically been modified by stock management, which under certain conditions has produced disturbance processes resulting in a dominant structure of native C3 grasses (Sinclair *et al.*, 2014; McIntyre *et al.*, 2022). Although *S. plana* is associated with a range of native grass species it has adopted the exotic weed Chilean Needle Grass (*Nassella neesiana*) as a novel food species (Richter *et al.*, 2013a).

S. plana is particularly threatened by the influx of exotic weed species, particularly invasive pasture grasses favoured by habitat modification (Kutt *et al.*, 2015), which fundamentally alter site structure (O'Dwyer & Attiwill 2000; Jellie *et al.*, 2014; Kutt *et al.*, 2015). Small, isolated urban populations are threatened by genetic factors (Clarke & O'Dwyer 2000) related to recruitment failure and lack of gene flow (Mata *et al.*, 2017). Managing a site to support *S. plana* requires the maintenance of an appropriately structured grass sward dominated by C3 food grass species (DEWHA 2009), which involves intensive biomass reduction approaches, including sheep grazing, ecological burning and slashing (ACT Government 2017c).

This review identifies a primary need for:

- Formal identification keys for *S. plana* pupae and the different instar stages of the species. Such a key would allow for more reliable surveying of potential *S. plana* habitat during suboptimal flight seasons, through pupal case identification, reducing the rate at which populations are lost to development.

Additional areas requiring quantified research include:

- The response of the species to ecological burning (DAWE 2021a) and to different timing, duration and intensities of grazing (Kutt *et al.*, 2015), particularly during the flight season.

- The genetic diversity of *S. plana* is overdue a detailed reanalysis (Clarke & O' Dwyer 2000) due to the abundance of new populations that have been discovered over recent decades (DAWE 2021a).

Introduction

The Golden Sun Moth (*Synemon plana*) Walker, 1854 is a Castniid moth endemic to the grassland habitats of southeast Australia (Douglas 2007). *S. plana* entered a pronounced decline during the latter half of the twentieth century, with dwindling records and multiple perceived population losses (Edwards 1993, cited in ACT Government 2017a). Following an increased interest in the last known *S. plana* populations in Victoria during the early 1990’ s, the species was found to persist at multiple sites across the state (DAWE 2021a). *S. plana* has since received concerted conservation efforts across its national range, which have included the creation of many grassland reserves specifically dedicated to its conservation, which include comparatively large areas such as the currently acquired sections of the designated 15,000 ha Western Grassland Reserve bordering Melbourne (DEPI 2013; Victorian National Parks Association 2020; DAWE 2021a). In 2020, the threat category of *S. plana* within the state of Victoria was revised from Endangered down to a new classification of Vulnerable, predominantly due to the extent of these new populations (DAWE 2021a; SWIFFT 2022). This literature review seeks to collate recent academic findings on *S. plana*, in conjunction with government literature, professional consulting reports, and the professional experience of land managers. This synthesis is intended to highlight knowledge gaps in the ecological management of *S. plana*, as well as identifying best practice solutions and future directions for research. This literature review was initiated by Trust for Nature, in association with the Golden Sun Moth Conservation Fund. The review introduces the subject species in accordance with the conventions of the International Code of Zoological Nomenclature. Synemon plana is thereafter abbreviated as “*S. plana*”, in accordance with standard Linnean conventions of nomenclature for animals.

Relevant literature was sourced through the La Trobe University Library, using the search term “*Synemon plana*”. Each publication was then used as a source of further reference material through the process of “snow-balling”. Most consulting reports and government publications were publicly available from Internet searches. A minority of unpublished reports were sourced through requests to various ecological consultancy firms, private landowners and government personnel. Interviews were either completed in person, over the telephone, via conference software, including Zoom and Microsoft Teams, or as emailed communication. Land managers of council and state reserves, as well as private property owners with documented *S. plana* populations were interviewed for their perspectives on effective management of *S. plana* habitat. Most interviews were open-ended and did not follow a set list of questions. Some email communication with interview subjects was more structured in format, and respondents were asked a series of set questions for specific details about the properties that they managed and the challenges and successes that they had encountered as managers of *S. plana* sites.

Chapter 1. Biology and description of *Synemon plana*

1.1 General description

The Golden Sun Moth (*Synemon plana*) Walker, 1854, is a diurnal moth belonging to the cryptic and poorly known Castniidae family. In appearance *S. plana* has dark brown forewings, with fine greyish white markings, distinctive pale green eyes and butterfly-like club-tipped antennae characteristic of this family (Creagh 1991; Douglas 2004; Kallies *et al.*, 2020). Adults of the species lack functional mouthparts and are unable to drink or feed as a result (Douglas 2004, Richter *et al.*, 2013a). *S. plana* typically live for one or two days as an adult moth, although under cool weather conditions adults of the species have been recorded surviving for as long as five days (Cook & Edwards 1994, cited in Gibson & New 2007; Richter 2010). Like many other Castniidae species, *S. plana* rests with its forewings arched over its abdomen, which likely allows it to conceal its brighter hindwings from predators (Douglas 2004).

Unique amongst the Castniidae, adult *S. plana* are notable for sexual dimorphism (Figure 1), which produces an associated suite of characteristic behaviours. Males are longer-winged (35 mm) and possess dark bronze-brown hindwings with faint darker spots. The semi-flightless females have 30 mm long forewings and large abdomens. Female *S. plana* possess distinctive yellow-orange hindwings with black spots and 10 mm-long ovipositors (Douglas 2004). The adults of both sexes appear much darker after recently emerging from their pupae (B. Tomkins, Friends of Mt Piper (FoMP) 2022, pers coms.). Prolonged flight through vegetation can cause older individuals to lose wing scales and to appear much paler and drabber as a result (B. Tomkins 2022, pers coms.).

1.2 Adult behaviour

Females are reluctant fliers even when disturbed (Figure 2, Clarke & O' Dwyer 2000). They are normally capable of short clumsy bursts of up to a few meters, though they have been observed to move distances of over 40 m (Gilmore, cited in Biosis 2013). After hatching, adult female *S. plana* generally rest motionlessly in a conspicuous spot on the ground, occasionally signalling flying males by rhythmically flashing their hindwings (Douglas 2004; DEWHA 2009; Richter *et al.*, 2013b). Males have been recorded approaching non-signalling females from 10 cm away, implying that the species may also engage short-range pheromone-based signalling (Douglas 2004).

Males fly in a characteristic zigzag motion, about one metre over the grassland (Richter *et al.*, 2013a) for sustained bursts of up to several hundred metres while patrolling for females (Richter *et al.*, 2013b). Multiple records show that male *S. plana* favour low-biomass habitats, including recently mown strips, actively avoiding dense higher-biomass grass patches such as those formed by Kangaroo Grass (*Themeda triandra*) (Gibson 2006, cited in Gilmore *et al.*, 2008), typically not travelling more than 50 m into dense, higher-biomass vegetation of this kind (Edwards 1994, cited in ACT Government 2020a).

Mating typically lasts for up to five minutes (Clarke & O'Dwyer 2000; Gibson 2006) with only the fittest individuals of the larger adult population mating successfully. Once mating is complete, males typically resume patrolling for additional mates (Clarke & O'Dwyer 2000; Gibson 2006), while mated females begin walking and/or flying among the bases of grass tussocks in search of oviposition sites, during the afternoon (Richter *et al.*, 2013a). Sometimes however, females will remain stationary for extended periods after mating (Gibson 2006). As the female crawls along, she probes the soil at the base of tussocks for suitable host plants (Edwards 1997, cited in Douglas 2004). Females spend at least 40 minutes laying between 80 and 200 eggs, each slightly larger than 2 mm in diameter (Richter *et al.*, 2013a). She deposits these eggs at the base of tussocks growing within several metres of the mating site (Edwards 1994, cited in Richter *et al.* 2013a and ACT Government 2017a; Gibson 2006). Females are more capable of flying after depositing some of their egg load, which likely aids them to disperse the remaining eggs further around the site (Douglas 2004).

Figure 1. Female and male *Synemon plana* (Photo credits G. Jury and M. Scicluna respectively)

Figure 2. Female *S. plana* with characteristically large abdomen (Photo credit L. Gibson 2022)

1.3 Physiology and weather cues

Adult moths emerge for a short breeding period in the late spring-summer. They emerge under optimum weather conditions, peaking between the months of November through to early January (Kutt *et al.*, 2015). Local climactic and topographic conditions can alter the onset and duration of this flight period (Richter *et al.*, 2013a). In particularly warm regions of its range, such as in the Wimmera, *S. plana* emerges in late October and completes its season by mid-November (Douglas 1989-2003, cited in Douglas 2004). This early onset to emergence is due to the ground drying out earlier in the year at the far western edge of the species' range (T. Wills 2022, pers. coms.). Warm, dry spring weather prompts a similar late-October start to the season in the ACT (Clarke & O'Dwyer 2000). Conversely, wet spring and summer weather can cause the flight season to be delayed, with the first emergence occurring in December and then continuing until the end of January (Richter *et al.*, 2009; DES cited in Sugarloaf Pipeline Alliance 2011). The species has been recorded flying in Melbourne as late as mid-February (Ecology Australia, cited in Sugarloaf Pipeline Alliance 2011).

Adult *S. plana* rely upon inter-tussock spaces for basking habitat in order to sufficiently raise their body temperature to begin sustained flight (DAWE 2021a). Normally, *S. plana* become active when morning temperatures reach 20°C, provided wind and cloud cover are very low. *S. plana* have been observed climbing up grass tillers to aid in drying their body after emerging from pupae (B. Tomkins, FoMP, 2022, pers coms.). Males subsequently fly from late morning until around 2 pm (Gibson & New 2007; Greenville *et al.*, 2012; Kutt *et al.*, 2015) with flight sometimes continuing until 3 pm (Richter *et al.*, 2013b). Richter *et al.* (2013a) consider the species to favour a peak within this daily activity time; spanning from 11am–2pm.

More recent survey work has often detected *S. plana* both earlier and later in the day than this, however, flight observations are uncommon below 22°C across the southern edge of the species' range (Biosis 2018a). Accordingly, in southern Victoria, 22°C is a more reliable predictor for *S. plana* flight activity (New *et al.*, 2007), however, *S. plana* flight has still occasionally been documented in southern Victoria at temperatures as low as 19°C (Brown *et al.*, 2012). Cooler weather causes *S. plana* to become sluggish and apparently lowers their total daily flight capacity (DEWHA 2009). Flight activity may accordingly be altered by subtle environmental impacts, such as thick smoke blocking the sunlight over a grassland (Gilmore *et al.*, 2008). Less than 25% cloud cover is considered optimal flight weather for the species, however, high numbers of flying males have been recorded at 50% cloud cover and there have even been occasional records of multiple males flying under 75% cloud cover (Brown *et al.*, 2012). Biosis (2018) surveys have frequently observed *S. plana* during cloudy conditions like this. Wet weather also appears to hamper emergence of the species and during a particularly wet season, recorded numbers generally stay low throughout the entire season (DES cited in Sugarloaf Pipeline Alliance 2011).

Females remain active late into the afternoon while ovipositing, having been recorded still active until approximately 5 pm (Richter *et al.*, 2013b). In particularly hot weather (35-38°C), both females (Rowell, cited in ACT Government 2017a) and males (Clarke & Dear 1998) perch higher up on grass canes, probably to avoid the heat of the soil surface. Males tend to become less active at temperatures above 35°C (Gibson & New 2007) and mortality from heat stress has been implied at 38°C (Clarke & Dear 1998). However, males have been recorded flying at temperatures as high as 39°C (Gibson & New 2007).

Large numbers of flying males have nevertheless been sporadically observed as late as 5.30 pm (Biosis 2013; Taylor, private land manager, 2022, pers. coms). Wind speed, in combination with temperature, forms an additional driver of activity in *S. plana*, with typical conditions involving very light levels of wind. Male flight is still occasionally observed during moderate winds (gusting up to 27.6 km/hr), particularly during high temperatures (Brown *et al.*, 2012). Biosis (2018) have made multiple records of *S. plana* activity during both moderate and strong winds, including during cool, cloudy weather. During windy conditions male *S. plana* shelter on the lee side of ridges and dry-stone walls, where they often aggregate in groups (Rowell, cited in ACT Government 2017a; Backstrom & Forbes 2019). The dispersal capacity of male *S. plana* is considerably higher in windy conditions (B. Tomkins, FoMP, 2022, pers coms.). Individual male *S. plana* have been found lodged in air conditioner vents on the top floor of four-story buildings (B. Tomkins 2022, pers coms.).

1.4 Life cycle

The cream-coloured *S. plana* larvae hatch after incubating for 3–4 weeks (Edwards 1993, cited in Richter *et al.*, 2013a; Enderby & Koehler 2004; Douglas 2004). The different stages of the larval lifecycle have not yet been formally described, but *S. plana* larvae have been observed to fall into three broad size ranges, based upon age (Richter *et al.*, 2013a). *S. plana* hatchlings feed upon plant tissue at the base of the culms of the surrounding tussock (Edwards 1993, cited in Richter *et al.*, 2013a; Enderby & Koehler 2004; Douglas 2004). This first instar appears to be negatively photo-tactic, and readily seeks cover in grass tussocks and soil cracks (Douglas 2004). *S. plana* remain as a hatchling for up to a year, growing to between six and 13 mm long, before entering the next stage of the *S. plana* lifecycle (Richter *et al.*, 2013a).

Upon reaching a larger size (13–20 mm), larvae dig short, silk-lined tunnels into the soil and their diet shifts to the roots of their host plants. The larvae burrow out along rhizomes to produce a network of tunnels (Edwards 1993, cited in Richter *et al.* 2013; DSE 2004). Particularly large tussocks can potentially provide a larva with all the food resources that it needs to complete its underground metamorphosis. Travelling laterally through the soil in search of additional nutrition is implied to be a risky endeavour for larvae, as doing this expends energy without the guarantee of a food reward, particularly in sites where the food plants grow in low density (Edwards 1994, cited in ACT Government 2017c). The late instar larva can reach 20 mm long and develops a red-brown head capsule. *S. plana* larvae have been recorded to live for 3 years (Richter *et al.*, 2013a), although their maximum lifespan is potentially longer than this (Edwards 1994, cited in ACT Government 2017c). However, if conditions are perfect, *S. plana* may be capable of completing its larval lifecycle within a single year (Richter *et al.* 2013a; Kutt *et al.* 2015). Day length, soil temperature and the quality of the available food resources are all likely to provide triggers that determine whether a larva continues feeding, growing and developing, or ceases and instead enters annual diapause (Friberg *et al.*, 2012, cited in DAWE 2021a). These annual larval cohorts may be mostly isolated from each other, instead producing sequentially emerging moth cohorts that are temporally separated and have no ability to meet and breed.

Figure 3. *S. plana* pupal casings.

Photo credit, T. Wills & GHD Consulting 2022

This process could plausibly generate separate breeding populations that all occupy the same site, but emerge to breed on alternating years. Minor gene flow might still sometimes be possible due to diapausing individuals, which would shift into the following year’ s annual cohort in response to weather-based triggers (Clarke & O’ Dwyer 2000; Enderby & Koehler 2006; Richter *et al.*, 2013a; Kutt *et al.*, 2015). It is possible that *S. plana* are further structured within each annual cohort, with potential mating partners restricted to those that pupated at a similar time.

In the spring, the late instar larva digs a silk-lined tunnel back to the surface, where it produces a reddish-brown pupa (Figure 3, Richter *et al.*, 2013b). Typically, pupation is complete within six weeks, although there are records of sequestered *S. plana* pupae surviving in the soil for multiple years (DSE 2004; DEWHA 2009). Female pupae grow to larger sizes than male ones, a trait that allows trained experts to sex-ID collected pupal cases (Richter 2010, cited in ACT Govt 2017). Under suitable conditions, the adult moths continuously emerge from their pupae, across the mating season (Clarke & O’ Dwyer 2000; DSE cited in Sugarloaf Pipeline Alliance 2011).

There is a high male bias amongst the adult moths that hatch at numerous sites in the ACT, and this is likely to be representative of the species’ typical population structure throughout its broader habitat range (Richter *et al.*, 2013). Female *S. plana* hatch with fully developed eggs, seeking males for fertilization (Dear 1997). It is plausible that the species is also capable of limited supplementary parthenogenesis, although preliminary analysis suggests this not to be the case (Clarke & O’ Dwyer 2000; Douglas 2004). The intensive competition for mates among male *S. plana* has been hypothesised to selectively advantage those that can seek out freshly hatched, virgin females, using olfactory cues before they have begun to display (Douglas 2004).

Chapter 2. Current distribution and habitat occupancy

2.1 Population and total habitat occupancy

Australia' s total extant *S. plana* population is projected to comprise at least 10 000 adults across a given flying season, assuming optimal weather conditions. Its total habitat currently comprises a projected 141 472 km^2, of which it directly occupies a known 1572 km^2 (DAWE 2021a). *S. plana* is currently known from 164 sites (NSW Government 2022). The cryptic life cycle of this species nevertheless renders precise population estimates and comparisons between local populations fraught with difficulties (Gibson & New 2007). The Department of Agriculture, Water and Environment (DAWE 2021a) considers accurate determination of the true extent of potential suitable *S. plana* habitat to be a research priority for this species. Intensifying survey effort has revealed an abundance of *S. plana* populations occupying modified landscapes (New 2019). This has resulted in increasing scepticism among stakeholders regarding *S. plana'* s conservation status (New 2019).

2.2 Distribution

S. plana is believed to have originated in central Australia, before radiating through Victoria, and, more recently, into New South Wales (NSW) and the Australian Capital Territory (ACT) (Clarke & Whyte 2003) (Appendix A). In Victoria, *S. plana* occurs across the western Victorian Plains to the western and northern reaches of Melbourne, ranging north into Central Victoria and across the Central Uplands Bioregion (O' Dwyer & Attiwell 2000; Brown *et al.* 2012; Richter *et al.*, 2013b; Kutt *et al.*, 2015; Mata *et al.*, 2017). As of 2020, *S. plana* has been recorded at over 100 Victorian sites, with then-current data directly confirming its continued presence at 36 of these (ACT Government 2017; Craigie 2020 & Douglas 2020, *cited in* DAWE 2021a). *S. plana* also occurs in parts of NSW, where it is listed as Vulnerable (NSW Government 2022) and the ACT, where it is locally endangered (Mata *et al.*, 2017). Historical records show that the northern distribution of *S. plana* in NSW once included Winburndale, near Bathurst, and the Yass Plains (Richter *et al.*, 2009). It is known from 78 sites in the ACT and 48 sites in NSW (ACT Government 2017, OEH 2012, *cited in* DAWE 2021a). The species is locally extinct across large parts of its former range (Clarke and O' Dwyer 2000; Braby and Dunford 2006; Richter *et al.*, 2013a). Its western limit was formerly Bordertown in South Australia, though it is now presumed to be extinct in the state (Edwards 2004, *cited in* Douglas 2004 & DEWHA 2009). Data regarding the extent of its current range in NSW, as well as across some parts of Victoria, remains incomplete (DAWE 2021a). Accordingly, multiple undetected sites likely remain in these landscapes (Figure 4., DAWE 2021a).

During early European settlement, *S. plana* was common and widely distributed across the grassland habitats of south-eastern Australia. Specimens were collected from a range of sites in the 1870' s, including at Bathurst and Bordertown (Creagh 1991). Museum records indicate that the species entered a rapid decline over the latter half of the twentieth century (Edwards 1993, cited in ACT

Government 2020a). By 1991, *S. plana* was known only from a single site in Victoria, and several grasslands in the ACT (Creagh 1991). Knowledge on *S. plana* was scant at this time, and the species still lacked a widely accepted common name (Creagh 1991). Over the next three years, an additional population of *S. plana* was discovered at a site in NSW, as well as a further four sites in Victoria, including at Nhill in the Wimmera; where it was found to coexist with a critically endangered morph of *S. selene*, the Pale Sun Moth (Douglas 2003). *S. plana* was discovered in Melbourne in 2003, when searches of proposed development sites identified many surviving populations in the north and west of the city (Gilmore *et al.*, 2008). One of these new sites, Craigieburn Grassland Reserve, comprised one of the biggest Victorian Volcanic Plains (VVP) grassland remnants known (Appendix E1). This site was the largest *S. plana* habitat in the state, discovered at that time (Gibson & New 2005). Surveys completed from 1994–2009 discovered a total of 40 additional Victorian populations, including the new Melbourne sites, along with a further 47 sites in NSW, and a further 20 in the ACT (Richter *et al.*, 2009). As more thorough state-wide survey work ensued in Victoria, the number of known populations almost doubled between the years of 2007 and 2012 (Brown *et al.*, 2012). However, most of these newly discovered populations occupied land fragments less than 5 hectares in size, that were at risk of both habitat disturbance and development pressure from ongoing urban and agricultural sprawl (Braby and Dunford 2006; Gibson and New 2007; Gilmore *et al.*, 2008; Richter *et al.*, 2013).

2.3 Habitat preference

2.3.1 Habitat structure and landscape factors

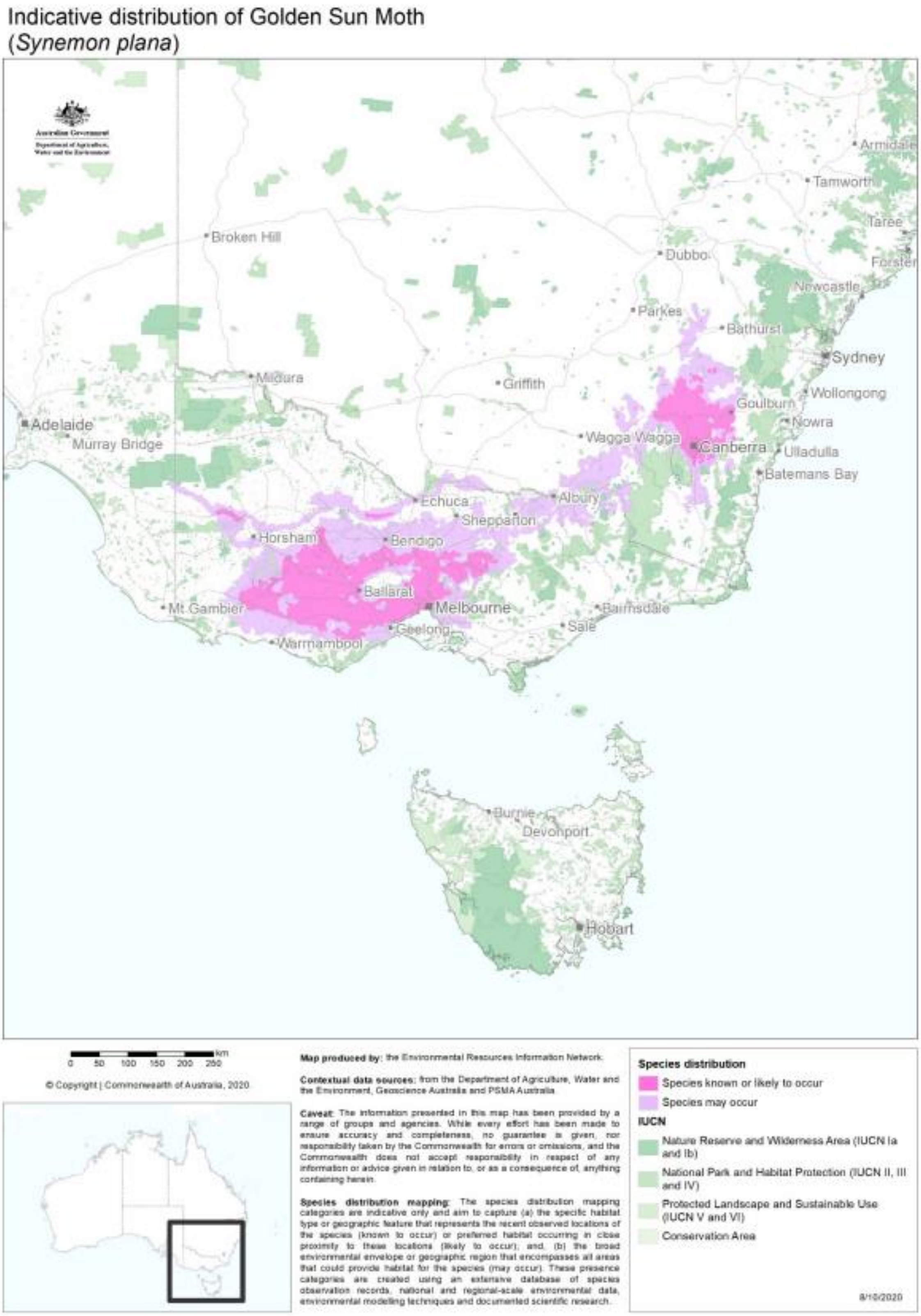
S. plana preferentially occupies temperate C3 grasslands, dominated by wallaby grass (*Rytidosperma* spp.) and spear-grass (*Austrostipa* spp.), favouring areas of low biomass with suitable inter-tussock spacing (Kutt *et al.*, 2016). While the exact species assemblages and percentage covers of grasses vary heavily between occupied sites, *S. plana* abundance is positively correlated with combined species richness amongst *Rytidosperma* and *Austrostipa* (Kutt *et al.*, 2016). Increases and declines in the richness and abundance of these genera correspond to associated rises and falls in recorded *S. plana* numbers from year to year (Kutt *et al.*, 2016).

Drought stress, associated with both the rain shadow effect (DSE 2009) and clay soils (Sinclair & Atchison 2012) have produced naturally treeless grassland landscapes in Victoria that are associated with *S. plana*. The historic tree-less state of these landscapes was likely also influenced by lower atmospheric CO_2 levels, which would have reduced growth and recruitment rates in trees and shrubs, further buffering the established grasslands from invasion by woody species (Berry & Roderick 2005). Historically, the habitat structure within such areas was partly driven by macropod and other native herbivore grazing, although periodic, intense fire events (wildfires and cultural burns) are likely to be the primary deep-time driver responsible (Dorrough *et al.*, 2004; Mata *et al.*, 2017). Many areas currently inhabited by *S. plana* shifted from dominance by *T. triandra* to C3 grass dominance, as a result of agricultural grazing pressure across the past 200 years (Sinclair *et al.*, 2014; Morgan *et al.*, 2017; Morgan & Salmon 2019). Research has similarly shown that repeated slashing tends to increase the species richness of the grasses in C3 Grasslands by reducing the dominance of *T. triandra*

(Brawata *et al.*, 2017). Temperate grasslands accordingly form a shifting mosaic of C4 and C3 dominated areas, that change in abundance in response to a broad range of biotic and abiotic factors (Xie *et al.*, 2022). While this transition occurs naturally, it can also be exacerbated by anthropogenic land use.

The role of fire in C3-dominated grasslands is poorly understood (Brawata *et al.*, 2017). *S. plana* populations appear somewhat resilient to grassland fire events, but the ideal timing, temperature and frequency of fire in their habitat requires additional research (Edwards cited in DEWHA 2009; E. Stone, Hume City Council (HCC) 2022, pers. coms., also see Bainbridge & Longmore 2015). Some observations imply that *S. plana* populations initially drop after ecological burns, before rapidly recovering in numbers due to successful recruitment in the favourable site conditions produced (Brawata *et al.*, 2017). The response of C3 Grasslands to fire is highly sensitive to a range of different factors. High tussock mortality can arise under inappropriate frequency, temperature or timing of burns, which necessitates a vastly different and much more targeted fire regime than is normally suitable for *T. triandra* grasslands (Brawata *et al.*, 2017; Morgan & Salmon 2019).

Richter *et al.* (2013b) associate *S. plana* populations within the ACT with habitat that retains a high diversity of native forbs, noting that this generally implies an absence of past anthropogenic disturbance. These C3-dominated ecosystems generally occur at higher altitudes (DAWE 2020a) and are likely to retain many grazing sensitive species because, unlike most lowland *S. plana* habitat, their structure is maintained by abiotic factors (Wong & Morgan 2007). The processes responsible for maintaining a low-disturbance, C3-dominated grassland include topographic factors (DSE 2009; Richter *et al.*, 2013b), climactic conditions and rainfall rates (Kutt *et al.*, 2016; Xie *et al.*, 2022). In precolonial history, the lower atmospheric CO2 and higher rainfall, during the cooler months of the year would have favoured the physiology of C3 grasses over that of C4 grasses (Xie *et al.*, 2022). In the ACT, some of these stable C3 grasslands occupy frost hollows (Dear 1997); where icy conditions both stunt the growth of key woody species, and destroy their germinants, which allows grasses to remain dominant (Dy & Payette 2007), particularly the more frost tolerant C3 species (Marshall 2013). However, these areas are often dominated by Tussock Grasses (*Poa* spp.) (J. Morgan 2022, pers coms.), which appear unsuitable as *S. plana* habitat (see Bainbridge & North 2007).



Map produced by: the Environmental Resources Information Network.

Contextual data sources: from the Department of Agriculture, Water and the Environment, Geoscience Australia and PSMA Australia.

Caveat: The information presented in this map has been provided by a range of groups and agencies. While every effort has been made to ensure accuracy and completeness, no guarantee is given, nor responsibility taken by the Commonwealth for errors or omissions, and the Commonwealth does not accept responsibility in respect of any information or advice given in relation to, or as a consequence of, anything containing herein.

Species distribution mapping: The species distribution mapping categories are indicative only and aim to capture (a) the specific habitat type or geographic feature that represents the recent observed locations of the species (known to occur) or preferred habitat occurring in close proximity to these locations (likely to occur); and (b) the broad environmental envelope or geographic region that encompasses all areas that could provide habitat for the species (may occur). These presence categories are created using an extensive database of species observation records, national and regional-scale environmental data, environmental modelling techniques and documented scientific research.

Source: Base map Geoscience Australia; species distribution data [Species of National Environmental Significance](#) database

Threatened Species Scientific Committee

Figure 4. An Indicative Distribution of Golden Sun Moth (*Synemon plana*) (DAWE 2021a). TREC holds no rights to reproduce this image, and it is copied here as a placeholder out of completion.

Typically, *S. plana* habitat includes at least ten percent total cover of *Rytidosperma* species (Brown *et al.*, 2012), although recent surveys have recorded them on sites where cover is as low as three percent, particularly around stony rises (Biosis 2005, cited in Enderby & Koehler 2006; Rowell 2009; DAWE 2021a; Bainbridge *et al.*, 2006, cited in Urlus 2021) or indeed absent entirely within areas that are dominated by Chilean Needle Grass (*Nassella neesiana*) (Gilmore *et al.*, 2008; Richter *et al.*, 2009; Brown *et al.*, 2012). The overall size of individual tussocks appears to be an additional factor determining habitat suitability, with a study in the ACT observing higher numbers of *S. plana* at sites where the tussocks were larger in size, but sparser in overall distribution, with greater inter-tussock spacing (DAWE 2021a). *S. plana* is believed to favour these larger tussocks as laying sites because their increased root mass provides more nutrition for larval development (Rowell 2009; ACT Government 2017a). Additionally, these larger and more established tussocks are much less vulnerable to fire (Morgan & Salman 2019), so may arise as a more typical growth habit under certain biomass regimes. A study in central Victoria found that *S. plana* tends to favour areas where grass species are lower to the ground (Kutt *et al.*, 2016). Broader accounts have also observed its abundance in heavily mown areas (Rowell 2010, cited in ACT Government 2017c; Biosis 2017). Inter-tussock spacing, with areas of bare ground is likely to be important for various stages of the *S. plana* lifecycle and this appears to be particularly essential for mating (Mata *et al.*, 2017). Invasive grasses increase in biomass after heavy rain, which has been suggested to swamp pupae and to reduce the number of emerging adults at a site (Kutt *et al.*, 2015). As this species appears to favour different conditions depending on changing interannual weather patterns, it is important not be either too reductive or too prescriptive when it comes to assigning a given site's potential value as *S. plana* habitat (A. Kutt 2022, pers. coms.).

S. plana's habitat also encompasses grassy woodlands (O'Dwyer & Attiwill 2000; Gilmore *et al.*, 2008; Kutt *et al.*, 2014). Within woodland habitats, *S. plana* tends to be confined to large clearings, and it generally occurs in lower numbers than populations that occupy remnant native grassland (Hogg 2010 cited in ACT Government 2017a; ACT Government 2017c; DAWE 2021a). The species also tends to be less abundant within secondary grasslands created through tree removal (DAWE 2021a). Accordingly, both these habitat types are implied to be suboptimal peripheral habitat, which the moths expand into from core areas of grassland habitat (Hogg 2010, cited in ACT Government 2017a). Large populations have nevertheless been recorded occupying such areas (Dear 1997; DSE 2004).

Populations in Victoria typically occur between 95 m and 406 m above sea level, while those in the ACT and NSW normally occur at higher elevation; between 480 m and 720 m above sea level (Clarke & Dear 1998; DAWE 2020). The higher altitudes of the Great Dividing Range have been proposed as a hard barrier to *S. plana* dispersal (Clarke & Dear 1998). In general, *S. plana* favours dry soils (Kutt *et al.*, 2015; Bainbridge & Longmore 2016). At the landscape scale, the position of a given grassland will further determine its soil moisture levels and annual rainfall, which are both broader determinants of its suitability as *S. plana* habitat (DSE 2004; DEWHA 2009; Brown *et al.*, 2012; Kutt *et al.*, 2015). *S. plana* occurs on sites with a broad variety of soil types, ranging from clays (Mount Piper, Victoria) to loams (ACT) to sands (Salisbury, Victoria). Soil phosphorous levels have been proposed to restrict *S. plana* habitat occupancy at the landscape scale (O' Dwyer & Attiwill 1999). The soils of native grassland landscapes in Australia are naturally low in phosphorous (Morgan & Williams, cited in Brawata *et al.*, 2017, Mitchell *et al.*, 2019), which the species prefers. Soils with a high phosphorous content favour invasive annual grasses over *S. plana*' s native food plants (O' Dwyer & Attiwill 1999). Notably, *Rytidosperma* species are not restricted by the phosphorous (or indeed nitrate) levels themselves, but by the competition from exotic grasses that they engender (Mitchell *et al.*, 2019). Additionally, phosphorous, at high concentrations, is toxic to many soil invertebrates, which may be true for *S. plana* larvae (O' Dwyer & Attiwill 1999).

Adult moths will move around a grassland in order to maximise optimum habitat, with populations expanding into previously wet areas in dry seasons and contracting into the driest areas in wet seasons (T. Wills & A. Kutt, 2022, pers. coms). North facing slopes and raised areas act as critically important refugia for the species during wet seasons, however, the species does not seem to utilise exposed areas on hilltops in such a manner (T. Wills & A. Kutt, 2022, pers. coms.; Gibson 2006). High numbers of *S. plana* have also been recorded in "saddles"; grassy dips in the lee of two rock knolls, where adults appear to benefit from wind shelter provided by the surrounding topography (Bainbridge & North 2007). *S. plana* generally favours sites with a slight slope of less than 5°, particularly those facing north, although some populations do occupy flat sites (DAWE 2021a). Even within a single season, *S. plana* will often emerge patchily from different microhabitats across a site, at different times, as the local soil temperatures change (New *et al.*, 2007).

2.3.2 Food plants

S. plana is considered to particularly favour Short Wallaby Grass (*R. carphoides*) (Edwards 1993, cited in Richter *et al.*, 2013b), Lobed Wallaby Grass (*R. auriculatum*) (Rowell 2013), Bristly Wallaby Grass (*R. setaceum*) and Hill Wallaby Grass (*R. erianthum*) (O' Dwyer & Attiwill 2000). It has also regularly been recorded among Common Wallaby Grass (*R. caespitosum*) (Rowell 2009) and is further associated with Smooth Wallaby Grass (*R. laeve*) (ACT Government 2017c), Clustered Wallaby Grass (*R. racemosum*) (O' Dwyer & Attiwill 1999, cited in O' dwyer & Attiwill 2000) and Brown-back Wallaby Grass (*R. duttonianum*) (Biosis 2017, cited in Biosis 2018b). Rowell considers grassland communities that retain low-growing and more vulnerable *Rytidosperma* species, such as *R. carphoides* and *R. auriculatum*, to comprise superior habitat than communities that are dominated by disturbance-tolerant species, like *R. caespitosum* and *R. racemosum*. A survey at an *R. caespitosum*-dominated site in the ACT nevertheless produced higher counts, both of flying males and pupal shells, than were present at a similar *R. carphoides*-dominated site. This discrepancy was largely attributed to the low tussock density and generally larger plant size at the former site (7 tussocks/m² vs. 23 tussocks/ m²) (Rowell 2009). Rough Speargrass (*Austrostipa scabra*) is considered to be an *S. plana* food plant in Wimmera grasslands (Douglas cited in Braby & Dunford 2006) and the moth likely feeds upon Kneed Speargrass (*A. bigeniculata*) within more general native grassland habitat (Braby & Dunford 2006). *S. plana* is further associated with Veined Speargrass (*A. rudis*) (Biosis 2017, cited in Biosis 2018b).

There have been several observations of pupal cases found in Red-legged Grass (*Bothriochloa macra*) patches in the near absence of other native grasses, implying a potential role for this species as a food plant (Braby & Dunford 2006). Several populations have likewise been documented in areas dominated by Kangaroo Grass (*Themeda triandra*) (Enderby & Koehler 2006). *S. plana* has also been associated with Cane Speargrass (*Aristida ramosa*) (ACT Government 2017a). Richter examined the stable isotopes of gut contents extracted from S. plana larvae but could find no evidence of C4 grass consumption (2011, cited in ACT Government 2017a). C4 grasses were however rare in the associated study area, which renders this finding inconclusive (Osbourne pers. coms. 2015, cited in ACT Government 2017a).

The invasive Chilean Needle Grass (*N. neesiana*) is a common weed on many sites occupied by *S. plana*, with large populations of both adults and larvae recorded in grassland patches composed entirely of *N. neesiana* (Braby & Dunford 2006; Gilmore *et al.*, 2008; Richter *et al.*, 2013a, see Appendix E2). Observations of gravid females laying directly into *N. neesiana* tussocks, and abundant pupal casings at the bases of tussocks provide strong evidence that *S. plana* has opportunistically adopted this invasive stipoid grass as a novel food source. Richter *et al.* (2013a) have observed that larvae collected from *N. neesiana* dominated sites are significantly larger than specimens of the same age that had been collected from areas of native grassland where *N. neesiana* was absent. Furthermore, individual *N. neesiana* tussocks appear capable of supporting several larvae at once (SMEC 2014, 2015, cited in ACT Government 2017c). This has allowed *S. plana* to persist in disturbed and degraded grassland areas. *S. plana* can even spread into new habitats by moving along adjacent *N. neesiana*-dominated creeklines and roadsides, particularly where routine mowing keeps biomass low (Kutt *et al.*, 2015; ACT Government 2017a; E. Stone HCC 2022, pers. coms.). These peripheral areas around core *S. plana* sites are likely to provide important supplementary habitat, and potentially grant a limited degree of connectivity to otherwise isolated fragment populations (Kutt *et al.*, 2015; ACT Government 2017a). It is likely that the common ancestry between the South American *N. neesiana* and the Australian *Austrostipa* genus has led to the retention of homologous morphological traits between the two, which make *N. neesiana* a suitable host plant for *S. plana* (Richter *et al.*, 2013a). There are also concerns that *N. neesiana* feeding may facilitate localised natural selection within *S. plana* populations, leading to either reproductive isolation or loss of genetic variability (ACT Government 2017c; ACT Government 2020). Preliminary evidence suggests that *S. plana* may be able to feed on a related second species of invasive stipoid grass, the Serrated Tussock (*N. trichotoma*). Large numbers of *S. plana* larvae have been recorded among *N. trichotoma* root mass at a site in the ACT (DAWE 2021a; ACT Government 2017b). There may also be additional species of invasive grass with the growth habits and tissue structure needed to act as novel *S. plana* food plants (Brown *et al.*, 2012).

2.3.3 Patch size

In its natural state, *S. plana* is implied to form interconnected meta-populations occupying broad heterogeneous landscapes, with individual populations becoming isolated at distances greater than 400 m (Menéndez & Thomas 2000, cited in Kutt *et al.*, 2015). This functional connectivity requires access to dispersal corridors and favourable weather conditions (Kutt *et al.*, 2015). A population currently occupying a 0.51 ha habitat fragment at York Park, ACT has been monitored continuously since the mid-1990’s. Its ability to persist for more than 80 years under ongoing fragmentation and site disturbance indicates a degree of robustness to such processes within the species (Yeates cited in Umwelt 2020). The smallest patch size that can reliably contribute to the overall survival of the species is estimated to be 0.25 ha (DEWHA 2009). Large *S. plana* populations (>1000 individuals) can nevertheless persist in areas of under 400m^2 in size (Clarke & O’Dwyer 2000). As Biosis note (2018b), smaller sized *S. plana* habitat patches are best protected from edge effects if they are square or round in shape, in order to maximise the ratio of surface area to edge length. However, adult *S. plana* also persist within linear habitats including median strips, roadsides and rail sides, though typically adjacent to larger habitat patches (Crouch cited in DSE 2004; ACT Government 2017c; Umwelt 2020, B. Tomkins 2022, FoMP, pers coms.). Within the ACT, *S. plana* is considered to have expanded into linear habitats of this kind, through utilising *N. neesiana* as a novel food source (ACT Government 2017c).

Data from Sheoak, Yea, imply that small patches are vulnerable to inundation during wet seasons, which means that functional populations could require much larger areas in order to have the mixed topography they need in order to guarantee successful recruitment (Kutt *et al.*, 2015) (Figure 5). Reducing biomass in areas with higher soil moisture could somewhat offset this (T. Wills, 2022, pers. coms.; Richter *et al.* 2013a). Accordingly, habitat size may not necessarily present a major constraint to a population’s survival provided that it receives constant optimal management (Richter *et al.*, 2013a).

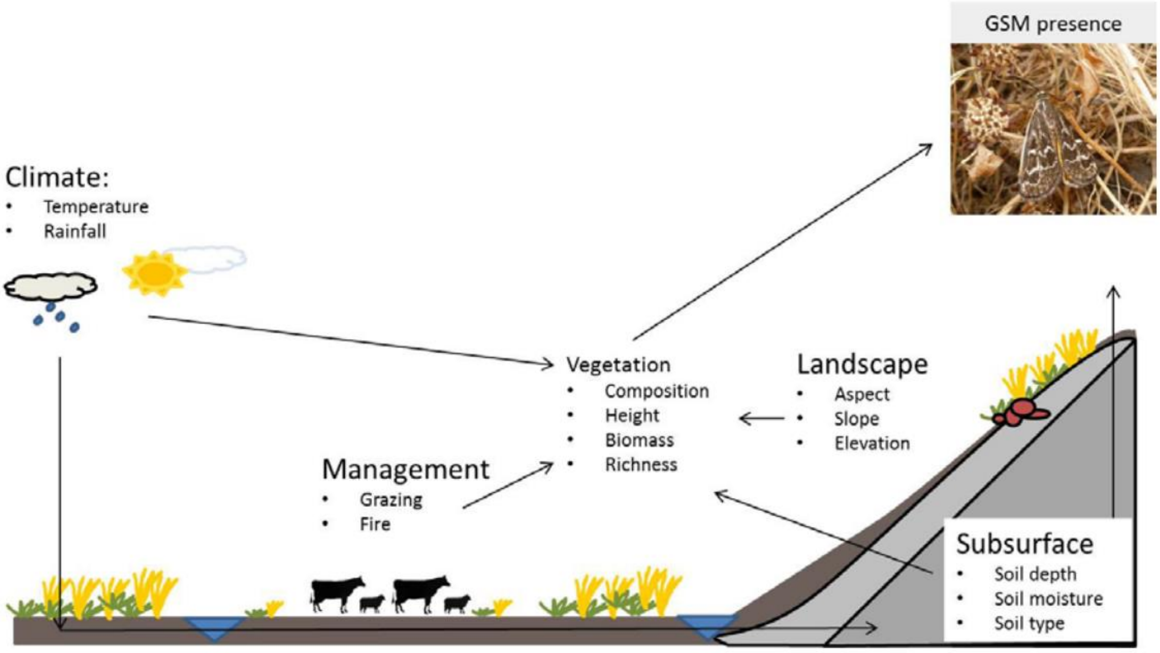


Figure 5. A conceptual Model of the Relationship between *S. plana* Ecology, Environment and Management (GHD 2013). Reproduced with the kind permission of Dr. Tim Wills and GHD Consulting, 2022.

2.4 Ecology of *Synemon plana*

2.4.1 Natural herbivory in *Synemon plana* habitat

In precolonial times the grasslands of south-eastern Australia were grazed by a faunal community that included the Eastern Grey Kangaroo (*Macropus giganteus*), the Emu (*Dromaius novaehollandiae*), the Common Wombat (*Vombatus ursinus*), various wallaby and bettong species, and a rich association of invertebrates (Fletcher 2006, cited in Brawata *et al.*, 2017). Vertebrate grazing was regulated by predation by Dingoes and the extinct Thylacine, as well as by Aboriginal hunting (Neave & Tanton 1989; Dorrough *et al.*, 2004; Antos & Williams 2015). This resulting population control would have accordingly produced a light and infrequent, but stable, grazing regime (Dorrough *et al.*, 2004), and an associated suite of co-evolved interactions between grazers and plant species (New 2019). Unlike modern-day hoofed agricultural herds, this native grazing community was made up of soft-footed species, which would not have compacted the soil heavily as they fed (Dorrough *et al.*, 2004). Grazing has been hypothesised to offset the competitiveness of dominant grasses, particularly taller species, which in turn increases species richness and favours perennial plants over annuals (Trémont 1994; Dorrough *et al.*, 2004). This allows *Rytidosperma* spp. to compete with the taller and generally dominant *Austrostipa* species (Douglas *et al.*, 2004). In general terms though, such structural impacts on *S. plana* habitat were likely minor, when compared with the landscape-scale impacts of fire (Lunt *et al.*, 2007).

M. giganteus is now the strongest and clearest driver of grassland biomass processes remaining from this native grazing community (Neave & Tanton 1989; Gott *et al.*, 2015). *M. giganteus* is an adaptable species capable of surviving in degraded and heavily modified landscapes where other vertebrate grazers have since been excluded (Antos & Williams 2015; Gott *et al.*, 2015). Kangaroos are accordingly "eco-system engineers" for temperate native grasslands, and their impacts on vegetation affect the broader resources available to a host of other species that occupy these landscapes (Howland *et al.*, 2014). Kangaroo mobs require trees for shelter and are reluctant to move long distances into open agricultural plains, from which all the savannah trees have been historically removed (Viggers & Hearn 2005). Intensive kangaroo grazing tends to keep the tussocks in a grassland at a short, uniform height, which remains throughout the whole year. Kangaroos favour *T. triandra* and trials have demonstrated that their grazing reduces its average tussock height from 21.1 cm down to 5-7 cm (Neave & Tanton 1989). Some sites occupied by *S. plana* are so heavily grazed by kangaroos that they are dominated by bare ground (Richter *et al.*, 2009). These conditions advantage faster growing C3 grass species over the normally dominant *T. triandra* and can produce a transitional shift to C3 dominance (Morgan *et al.*, 2017; Morgan & Salmon 2019; J. Morgan 2022, pers coms.). In contrast, dry, nutrient poor grasslands that are naturally dominated by *Austrostipa* and *Rytidosperma* species, such as occur on the Wimmera Plain and in the Victorian Riverina, do not respond favourably to grazing, and the biomass level in such grasslands appears, primarily, to be regulated by moisture levels (Schultz *et al.*, 2011).

Invertebrates comprise a major component of the grazing community in grassland environments, although their overall impacts are still poorly understood (Neave & Tanton 1989; Antos & Williams 2015; New 2019). This community includes lepidopteran larvae (Antos Williams 2015), such as the larvae of the Grass Anthelid (*Pterolocera amplicornis*), Toothed Anthelid (*Anthela denticulata*) and Eyespot Anthelid (*A. ocellata*) (New 2019; Herbison-Evans & Crossley 2022). The grazing insect community associated with grassland landscapes in south-eastern Australia is itself affected by vertebrate grazing and fire cycles (New 2019). Such relationships are dynamic, and vary at the landscape scale, according to geographic and climatic factors (New 2019). The ant communities associated with the root systems that *S. plana* larvae feed upon are the subject of current ongoing research (Yeates, Osbourne & Gibbons, cited in Umwelt 2020). Ant communities construct nests in fixed locations, from which they forage out for nutrients in a manner analogous to stoloniferous plants (Anderson 1995). Accordingly, examining this component of the sub-surface soil community may provide useful insights into the soil ecology that the species depends upon during the early stages of its development, particularly when its species composition changes due to disturbance processes (Yeates, Osbourne & Gibbons, cited in Umwelt 2020).

Chapter 3. Key gaps in knowledge

The cryptic nature of *S. plana* combined with its short adult lifespan and highly specific emergence conditions have confounded attempts to delineate the full extent of its current distribution (DAWE 2021a). Further, uncertainties regarding the full range of *S. plana*'s food plants and knowledge gaps associated with the early stages of the *S. plana* lifecycle, have added to ambiguities in survey results (Mata *et al.*, 2017, DAWE 2021a). *S. plana* habitat has been shown to be complex and heterogeneous, which creates difficulties in assigning optimal habitat values for its conservation (Kutt *et al.*, 2015). The management of small and fragmented *S. plana* populations is limited by a lack of knowledge regarding the natural sex ratios and mating biases that the species displays at the population level (Richter *et al.*, 2013a). There is also a lack of robust data on the resilience of *S. plana* to fire, which frequently leads to mandatory restrictions upon the available conservation actions that can be deployed at sites occupied by the species (DAWE 2021a).

3.1 Knowledge gaps

Despite decades of research, contemporary academic publications describe *S. plana* as "data deficient" (Mata *et al.*, 2017). The life cycle of this species confounds easy research and limits the accuracy of clear categorical answers regarding its environmental needs or how to provide for them (Gibson & New 2007). *S. plana* is an elusive animal with a cryptic lifecycle, a rapid turnover of individuals and extremely specific activation cues that can be difficult to predict (ACT Government 2017c). This makes its general population health and habitat occupancy difficult to gauge with confidence and renders assigning concrete population numbers fraught with difficulty (Gibson & New 2007; Kutt *et al.*, 2015). This elusiveness has led to a skewed impression of its scarcity, when in truth it is often simply under-surveyed and under-detected (New *et al.*, 2007; New *et al.*, 2012). Accordingly, the extent to which undetected populations occur at the broader landscape scale is one of the primary knowledge gaps in the understanding of this species. The paucity of integrated survey data or indeed, its complete absence in some situations is particularly pronounced across large areas of Victoria and New South Wales (DAWE 2021a). This is also true for sites where populations have been declared extinct; some of which were no doubt pre-emptively assigned (New *et al.*, 2007). These assessments derive from failure to identify the species because of varying weather patterns between years, changed biomass routines, inappropriate survey conditions and inactive populations being surveyed during diapause (New 2015). In truth, almost any area of grassland or grassy woodland within *S. plana*'s range that retains either some native grasses or *N. neesiana* has the potential to support *S. plana*, and even this projected habitat range has been assembled from insufficient data (Biosis 2013).

A comprehensive understanding of the food plants utilised by *S. plana* is likewise lacking. Several common native grasses can only be loosely associated with *S. plana* habitat, as only occasional anecdotal observations of oviposition or pupal case presence have been recorded (e.g., *Bothriochloa macra*, Braby & Dunford 2006; *Microlaena stipoides*, Gibson & New 2007). While these observations are useful proxies for more targeted data, there is a need for contained larval trials in order to establish whether such species are genuinely capable of providing food resources for *S. plana* (Gilmore *et al.*, 2008). Observations based on ovipositing alone are particularly ambiguous, as the females have been recorded to probe the soil with their ovipositors in search of suitable host plants (Edwards 1997, cited in Douglas 2004). Similarly, there have been no detailed studies into longer term survivorship of *S. plana* populations in *N. neesiana* grasslands (Kutt *et al.*, 2015). Further comparative larval studies exploring the potential physiological differences between *S. plana* populations that feed exclusively on *N. neesiana*, and those that feed on native grasses would be beneficial to these ends (Richter *et al.*, 2013a).

Attempts to quantify the optimal density and condition of food tussock plants around an annual cycle are highly subjective and tend to be overly prescriptive (Mata *et al.*, 2017; A. Kutt 2022, pers coms). This is compounded by the finding that *S. plana* has differing habitat requirements across the El Niño and La Niña cycle, which vary further at the microsite level based upon a complex interplay of biotic and abiotic factors (Brown *et al.*, 2012; Kutt *et al.*, 2015). Current understanding of the larval lifecycle is too poor to assign confidant management calls on appropriate biomass levels across varying conditions. Currently, information on the duration required for the species to grow to adulthood, its capacity to extend this, and the environmental cues involved, remain speculative only (New 2012; Mata *et al.*, 2017). These factors are even more tenuously understood with regards to longer-term climate change, and its impacts on the population ecology of this species (Kutt *et al.*, 2015). The immature stages of *S. plana* have not been formally categorised or described, which imposes constraints on understanding the lifecycle of the species at different sites (New 2012; Richter *et al.*, 2013b). The lack of accessible identification keys for pupae and larvae presents an obstacle to compiling supplementary data on population health and seasonal activity, which would aid site management aimed at conserving this species (see Richter *et al.*, 2013b).

There is very little information available on the fecundity of different *S. plana* populations (Richter *et al.*, 2013a), particularly with regards to minimum population size and the potential effects of inbreeding (Clarke & O'Dwyer 2000). The factors governing an individual's sex are not known for *S. plana*, nor are the causes behind the heavy male sex bias observed in Australian Capital Territory populations (Richter *et al.*, 2009; Richter *et al.*, 2013b). More detailed study of *S. plana* populations across the species' broader national distribution are needed in order to understand whether this sex bias is site-specific or a general trait exhibited by all *S. plana* colonies (Richter *et al.*, 2009).

The response of the species to the fire cycle is another key knowledge gap that is imperative for the effective management of this species, particularly at smaller sites that support other significant taxa. Detailed research into this area needs to draw upon current knowledge of *S. plana* ecology and to employ measurements such as breeding success, ranging behaviour, population abundance and individual mortality (DAWE 2021a). Some additional areas affecting the management and design of small reserves that are still lacking data include the impacts of litter, predation (Bainbridge & Longmore 2016) and herbicide (Mata *et al.*, 2017) on this species and its associated invertebrate community. In the case of predation, almost nothing is known on the impacts of invasive species to *S.*

plana, particularly those of invasive invertebrates (Yeates, Osbourne & Gibbons, cited in Umwelt 2020). All these factors can potentially play roles in how conservation reserves are designed and maintained, in order to best support *S. plana* and the broader ecological communities that it occurs within.

Chapter 4. Central and emerging threats

Museum records indicate that *S. plana* was still common and widespread prior to 1950, after which it began a marked decline, becoming extinct at multiple locations across its range (Edwards 1993, cited in ACT Government 2017a). Some of the key processes responsible relate to the flow-on effects of livestock grazing, pasture improvement and ploughing, each of which can lead to soil disturbance and increased invasion by exotic weeds, radically changing the biomass levels of a site (Richter 2010, cited in Act Government 2020a). The widespread addition of superphosphate to soils favoured influxes of exotic pasture grasses, which critically damaged the structure of the vegetative communities on which these populations depended (O' Dwyer & Attiwill 1999; Richter 2010, cited in Act Government 2020a; Gott *et al.*, 2015; Mitchell *et al.*, 2019). A suite of related changes to soil chemistry, water tables and nutrient cycling in these systems (particularly phosphorous and nitrogen), further drive the observed declines in *S. plana*' s population by facilitating the invasion and dominance of weeds (Australian Government 2013). The survival of *S. plana* across its national range is difficult to predict under a changing climate (Kutt *et al.*, 2015). Small, isolated urban populations are threatened at the local level by genetic factors (Clarke & O' Dwyer 2000), including genetic drift due to individual attrition from exotic predators (Mata *et al.*, 2017).

4.1 Habitat destruction

Post-colonial human activity has reduced the temperate grasslands and grassy woodlands of south-eastern Australia to less than 4% of their former range (Lunt *et al.*, 2007; Jellie *et al.*, 2014; Kutt *et al.*, 2016). This has resulted in a near 99% reduction of suitable *S. plana* habitat (DAWE 2020, cited in DAWE 2021a). These landscapes continue to face ongoing pressure from developers. Many extant populations occupy urban growth areas on the borders of cities (Gilmore *et al.*, 2008; Mata *et al.*, 2017). A large population on the western border of Melbourne was, for example, destroyed in 2008 in order to build the Melbourne wholesale Fruit and Vegetable market (Bainbridge & Longmore 2016). Populations on private land face uncertain futures in the absence of applied conservation management (Gilmore *et al.*, 2008). While many such areas have been converted to exotic grass dominance through pasture enrichment processes (O' Dwyer & Attiwill 1999; ACT Government 2020a), other sites have been cleared entirely with heavy machinery to produce broadacre cropping land (Clarke & O' Dwyer 2000; Biosis 2018b; Victorian National Parks Association 2020).

The loss of connectivity, compounded by edge effects, have produced severe biodiversity declines within these ecosystems. Fauna species reliant upon specific vegetative communities growing at optimal densities have been hit particularly hard (Dorrough *et al.*, 2012; Kutt *et al.*, 2015). The limited ability of female *S. plana* to disperse under fragmentation was one of the primary reasons for the species' original listing as Critically Endangered in Australia (DEWHA 2009). Populations that occupy isolated habitat fragments are particularly susceptible to site disturbance. The inability of new individuals to colonise available habitat renders the isolated survivors increasingly vulnerable to extinction (Clarke & Whyte, 2003). *S. plana* is unable to recolonise these empty grassland sites without access to suitable connecting habitat (Kutt *et al.*, 2015)

The habitat fragments that remain have experienced major changes, both to their natural fire cycles and to the intensity at which they are grazed, which has severely impacted their native biodiversity (Lunt *et al.*, 2007; Kutt *et al.*, 2015). Livestock grazing has altered the structure of many such landscapes, from originally being dominated by tall, native, perennial tussock species, to domination by short, winter-growing, exotic annual grasses (Dorrough *et al.*, 2004; Lunt 2007). Ploughing has also heavily altered the vegetative community of native grasslands (O' Dwyer & Attiwill 1999). Most of these ecological changes can now never be entirely reversed. Even restoring the full pre-colonial ecological function of a single site is a difficult task because many of the threatened forbs that are native to these landscapes lack soil seed reserves to recruit from (Morgan 1998; Lunt *et al.*, 2007). Annual grasses also alter nutrient cycles, locking in a cyclical process of transition away from the natural state of the ecosystem (Prober *et al.*, 2005, cited in Lunt *et al.*, 2007).

These changes have caused an associated reduction in the host plants utilised by *S. plana* (Richter *et al.*, 2013a). There have been severe losses to the invertebrate biodiversity of native grasslands as a result of this combined landscape-level modification (O' Dwyer & Attiwill 1999). Four monitored *S. plana* populations in the Australian Capital Territory (ACT) went extinct in the late 1990' s as a direct consequence of declining site quality (Clarke & Dear 1998; Clarke & Whyte 2003; Sharp 2009). At one such site, Yarramundi Grassland, this was specifically due to high biomass levels, weed invasion and associated reductions in food plants (Sharp 2009; ACT Government 2017c). Current estimates examining habitat area as an independent variable, suggest that *S. plana* populations occupying grassland areas smaller than 10 ha will be adversely affected by any significant modification of their remaining habitat. Populations in larger habitats are estimated to face adverse consequences if 0.5 ha or more should be modified at a time (DEWHA 2009).

4.2 Invasive plants

Invasion by exotic plants has been an ongoing threat to native grasslands throughout European colonial history (Lunt *et al.*, 2007). A typical plains grassland landscape that has experienced historic modification displays a characteristic absence of native floral species resulting from their displacement by exotic species (Australian Ecosystems 2019). *S. plana* is threatened by the conversion of its habitat into novel ecosystems dominated by invasive plants (Kutt *et al.*, 2015). A broad array of invasive weeds threatens the temperate native grasslands of south-eastern Australia, including perennial grasses (e.g., Perennial Ryegrass (*Lolium perenne*)), annual grasses (e.g., Wild Oats (*Avena* spp.)), broadleaves (e.g., St John' s Wort (*Hypericum perforatum*)) (DEWHA 2009; ACT Government 2017c) and woody weeds (e.g., Gorse (*Ulex europaeus*)). Fragmented grasslands are particularly at risk from exposure to weed incursions when adjacent external areas are modified (DEWHA 2009). This is likely to continue escalating into the future, potentially exacerbated by the effects of anthropogenic climate change, which could alter both the severity and the impacts of current invasive weeds and facilitate the establishment of additional exotic species (Richter *et al.*, 2013a).

S. plana populations that survive outright habitat destruction face the primary threat of the loss of their food plants and the typical habitat structure that these ecological communities depend upon (DAWE 2021a). This is primarily the result of ongoing anthropogenic pressures, which result in ongoing disturbance, edge effects and weed invasion (O' Dwyer & Attiwill 2000; DEWHA 2009; Kutt *et al.*, 2015), exacerbated by the desirability of these areas for urban and agricultural development (Richter *et al.*, 2013b; DAWE 2021a). Influxes of invasive pasture grasses are particularly damaging to the habitat structure of these landscapes. Seed from these grasses can spread on vehicles, or from grass cuttings left to accumulate on sites (Australian Government 2009). Removal of exotic grasses from *S. plana* habitat has accordingly been shown to provide direct increases to *S. plana* abundance at affected sites (O' Dwyer & Attiwill 2000).

Increased nutrients aid the weed invasion process, providing conditions in which exotic species can outcompete indigenous species (Lunt *et al.*, 2007). Pasture improvement programmes using superphosphate became increasingly prevalent from 1890 onwards and are responsible for a massive ongoing incursion of weedy grass species into existing grassland remnants (Gott *et al.*, 2015). Some of the most prevalent exotic species at managed *S. plana* sites are perennial C3 grasses. *L. perenne* is perhaps the most destructive of these and is particularly competitive in the soil conditions produced by past phosphorous treatments (O' Dwyer & Attiwill 1999; O' Dwyer & Attiwill 2000). Toowomba Canary-grass (*Phalaris aquatica*) and Cock' s Foot (*Dactylis glomerata*) are particularly prevalent weeds in *S. plana* habitat as well, able to readily outcompete desirable food grasses, and, in the case of *P. aquatica*, even *Nassella neesiana* (Richter *et al.*, 2009; Biosis 2018a; Umwelt 2020; Australian Ecosystems 2019). Additionally, Great Brome (*Bromus diandrus*), Soft Brome (*B. hordeaceus hordeaceus*), Yorkshire Fog (*Holcus lanatus*) (Douglas 2004; Parsons Brinckerhoff Australia 2008; Australian Ecosystems 2019), Brown Top Bent Grass (*Agrostis capillaris*) and Creeping Bent Grass (*A. stolonifera*) (Biosis 2018b) have emerged as dominant weedy grasses at certain sites. Annual C3 grasses proliferate in a similar manner and can be difficult to control due to their transitory growth cycle and tendency to alter soil chemistry through accumulating biomass in self-sustaining feedback loops (Lunt *et al.*, 2007; Wong & Morgan 2007). On sites managed for *S. plana* conservation, these species include Tall Fescue (*Festuca arundinacea*), Meadow Fescue (*F. pratensis*) (Parsons Brinckerhoff Australia 2008), Sweet Vernal (*Anthoxanthum odoratum*) (Bainbridge & Longmore 2015), Barley Grasses (*Hordeum spp.*); particularly Hare Barley (*H. murinum leporinum*), Wild Oats (*Avena* spp.); particularly Slender Wild Oat (*A. barbata*) and Common Wild Oat (*A. fatua*), Quaking Grasses (*Briza* spp.) and Fescues (*Vulpia* spp.) (DSE 2004; Douglas 2004; ACT Government 2017c; ; Biosis 2018b; Australian Ecosystems 2019).

Perennial exotic C4 grasses are especially competitive at locking up space in native grasslands and excluding native grass species, due to their hardiness and longevity (ARI 2021). Areas dominated by these grasses lack the inter-tussock spacing required by *S. plana*. These species are currently assumed to be unable to provide food resources for larvae (Richter *et al.*, 2009; Biosis 2018a; Australian Ecosystems 2019; Umwelt 2020;). African Lovegrass (*Eragrostis curvula*) is a C4 grass considered to be an emergent threat to *S. plana* sites, and is particularly competitive in dry, nutrient poor soils (ACT Government 2017c). Dallisgrass (*Paspalum dilatatum*) is also controlled to improve *S. plana* habitat in some areas (Parsons Brinckerhoff Australia 2008). Such species can readily degrade *S. plana* habitat, threatening whole populations, particularly at fragment sites (Richter *et al.*, 2013b).

The temperate grasslands occupied by *S. plana* are further threatened by a host of invasive broadleaf weeds, including St John's Wort (*Hypericum perforatum*) (Richter *et al.*, 2009), Variegated Thistle (*Silybum marianum*) (Biosis 2019b), Paterson's Curse (*Echium plantagineum*) and Ribwort Plantain (*Plantago lanceolata*) (Parsons Brinckerhoff Australia 2008). White Clover (*Trifolium repens*) is particularly damaging at some sites, where it vigorously outcompetes *Austrostipa* and *Rytidosperma* species (DEWHA 2009). Flat Weed (*Hypochoeris radicata*) and Cape Weed (*Arctotheca calendula*) often become prevalent in *S. plana* habitat because they are generally unaffected by grazing regimes (DSE 2004). In addition to historic site usage and edge effects, the prevalence of weed species within *S. plana* habitat depends heavily upon the existing management structures engaged at habitat remnants. Inappropriate site management can create conditions that favour exotics over desirable natives, in addition to a slew of other problems (DAWE 2021a).

4.3 Inappropriate management

Livestock grazing, pasture improvement and ploughing can all radically change the biomass levels of a site, leading to soil disturbance and increased invasion by exotic weeds (ACT Government 2020a). Fertilising soils with phosphorous favours annual grasses, which damages the structure of the vegetative community, and may poison *S. plana* larvae directly (O'Dwyer & Attiwill 1999; Douglas 2004). Similarly, changes to the water table, increased salinity and exposure of the soil to polluting chemicals are all potentially catastrophic for *S. plana* populations (Australian Government 2013). Pesticide application to pastures has never been studied in association with *S. plana* but is expected to be severely harmful to larvae (DSE 2004).

The resting state of the grasslands that *S. plana* occupies across much of its range is one of dominance by Kangaroo Grass (*Themeda triandra*), a species that readily deposits leaf litter as it grows (Stuwe & Parsons 1977). With too much accumulated biomass, the overall structure of the site can shift towards a tall, dense grass sward, that lacks inter-tussock spacing (ACT Government 2020a). Native grasses, like *T. triandra* can threaten *S. plana* if they are left to accumulate, resulting in dense biomass unsuitable for the species. *T. triandra* can accordingly dominate and suppress the grass species required by *S. plana* as food plants (Stuwe & Parsons 1977; Richter *et al.*, 2009). As current research stands, *T. triandra* is regarded as an inappropriate species for the restoration of degraded sites that are primarily intended for *S. plana* conservation (Umwelt 2020). In general, grass species that are known to climax in a dense, closed tussock sward, apart from proven *S. plana* food plants, should not be restored at fragment sites utilised directly for *S. plana* conservation (Parsons Brinckerhoff Australia 2008). Even desirable *Rytidosperma* species can form an unsuitable dense structure if appropriate biomass reduction measures are not in place (DSE 2004).

Conversely, a too intensive disturbance regime can reduce the vegetation on a site down to sparse stubble dominated by large patches of bare ground (ACT Government 2020a). This can result from overstocking or overgrazing an area (New *et al.*, 2007), overly intensive mowing, or heavy foot or vehicle traffic (DEWHA 2009). This kind of intensive disturbance causes soil compaction, which limits the root growth of native grasses, depriving *S. plana* larvae of food. The effects of soil compaction upon the early stages of the *S. plana* lifecycle are not well understood, however soil compaction likely exposes larvae and eggs to higher soil temperatures and a greater risk of desiccation (Douglas 2004; Act Government 2017c). Grazing by cattle is particularly likely to cause soil compaction, especially when soil conditions are wet, which leads to pugging (Greenwood & McKenzie 2001; Agriculture Victoria 2022). However, particularly dry grassland sites such as those in the Wimmera, are the least capable of recovering from such disturbance (Greenwood & McKenzie 2001). Mowing or slashing flowering or seeding native grasses can prevent the recruitment of new germinants (DEWHA 2009). Sites without adequate recruitment are more prone to erosion, desiccation and weed invasion, and can lose desirable food plant species entirely due to senescence (Lunt *et al.*, 2007; ACT Government 2020a; DAWE 2021a).

Targeting *N. neesiana* as a weed can prove dangerous to *S. plana* populations on small and isolated sites. In the ACT, some very large *S. plana* populations sustained critical reductions in numbers after their core food plant was eradicated by site management. These declining populations were then directly threatened with extinction (Richter *et al.*, 2013a). Other impacts resulting from herbicide use that are potentially relevant to *S. plana*, are poorly understood. *Rytidosperma* species are particularly susceptible to Glyphosate spray drift, so current recommendations advise against the use of this herbicide within areas set aside for *S. plana* conservation (DAWE 2021a).

Additionally, the broader vegetative structure of a site can be affected by inappropriate tree plantings, as well as the addition of large structures. These can shade out areas of a grassland altering the optimal soil temperature and moisture levels needed by the species at different stages of its lifecycle. Increased shade can also alter the composition of grass species present (DEWHA 2009; ACT Government 2020). Adding shrubs and trees to a grassland also provides habitat for avian predators of *S. plana*. Installing solid barriers across a site, such as sheet-metal fencing, can divide a population of *S. plana* preventing gene flow during the summer flight period (DEWHA 2009).

Ecological burns are an essential management tool in many native grassland landscapes in order to reduce biomass, trigger the seeding of fire-adapted plant species and facilitate recruitment of species dependent upon inter-tussock spacing (Lunt *et al.*, 2007; Wong & Morgan 2007; Morgan & Salman 2019). *S. plana* populations can, however, be adversely affected by either too frequent, or too high temperature burn regimes, particularly during the spring or summer; when most late instar larvae begin pupating (spring), and while adults emerge (summer) (ACT Government 2020a). C3 Grasslands are particularly difficult ecosystems to burn in a manner that produces desirable results and are notable for burning at higher temperatures than typically occur in *T. triandra*-dominated grasslands (Sinclair *et al.*, 2014; Morgan & Salman 2019). Moreover, current research (Sinclair *et al.*, 2014; Morgan & Salman 2019), has focused on *Austrostipa* dominated systems. A study of burn impacts on semi-arid grasslands with mixed *Rytidosperma* and *Austrostipa* dominance in the northern Riverina found little positive effects from reintroducing burns to these landscapes (Wong & Morgan 2012). This study concluded that the C3-dominated grasslands examined were composed of grazing-tolerant species that were maintained in these systems through sheep grazing (Wong & Morgan 2012). Burning a site may also cause subtle impacts on *S. plana* larvae by triggering their food tussocks to exhaust energy reserves stored in their root mass, in order to produce new leaves (Edwards 1994, cited in ACT Government 2018). Specific knowledge regarding the role of fire in *S. plana* habitat, however, remains extremely sparse, and detailed research into the response of this species to different fire regimes remains a pressing need (DAWE 2021a). A too hesitant approach to prescribed burning can impose risks to other threatened taxa in the grassland, that benefit from a regular fire cycle (Lunt *et al.*, 2007; E. Stone HCC, pers. coms. 2022). Conversely, a too heavy-handed approach can be inappropriate for species composition, risking unacceptable levels of mortality to *S. plana*, its food plants, and the broader ecological community to which they belong (Lunt *et al.*, 2007; Morgan & Salman 2019; ACT Government 2020a).

4.4 Climate change

The intensive modification of historic *S. plana* grasslands make the isolated fragments that remain increasingly vulnerable to further ecological changes. Grassland systems in south-eastern Australia have already lost much of their former systemic versatility. This previous versatility provided some resilience and would have aided these systems in responding to the impacts of anthropogenic climate change (Prober *et al.*, 2012, cited in Kutt *et al.*, 2015). *S. plana* relies upon specific weather conditions to establish the extent of the potential habitat across a site that it can use in any given year. Global environmental change Is likely to upset this balance, reducing the capacity of *S. plana* to respond to seasonal weather changes, particularly at isolated and fragmented sites (Warren *et al.*, 2001, cited in Kutt *et al.*, 2015). *S. plana*' s response to drought is an aspect of its ecology that requires further study (DEWHA 2009). Over the short-term, a drying climate could potentially benefit this species, by reducing both biomass levels, and inundation, on sites that it occupies (T. Wills, pers. coms. 2022). However, annual summer rainfall is predicted to increase across the northern portions of its range (CSIRO, BOM, cited by Xie, *et al.*, 2022). In general, the responses of Australia's grassland systems to anthropogenic climate change, and the resulting changes to annual rainfall patterns, are poorly understood (Xie, *et al.*, 2022), which further compounds the difficulty of predicting the future of *S. plana* under a changing climate. Small, isolated *S. plana* populations are vulnerable to grassfires, and would be impacted by the increased incidence of fire events (ACT Government 2020a). *S. plana* populations reliant upon *N. neesiana* may be impacted by the reduced drought-tolerance of this species, compared to *Rytidosperma* species (ACT Government 2019).

Within Victoria, it is likely that the projected reduction in annual rainfall will have large ecological impacts to grasslands, particularly those that exist under rain shadows (DSE 2009). Such areas will undergo pronounced structural changes if their annual rainfall should drop further, and potentially shift toward chenopod dominance (DSE 2009). C3 grasses are likely to face increasing competition from C4 grass species, due to projected annual reductions in cool season rains (CSIRO, cited by Xie *et al.*, 2022). Rising atmospheric CO_2 levels are likely to somewhat offset this, because this change would advantage the photosynthetic process engaged by C3 grasses (Morgan *et al.*, 2011, cited in Xie *et al.*, 2022). Certain weed species are likely to proliferate under hotter and drier conditions, such as African Lovegrass (*Eragrostis curvula*) (Richter *et al.*, 2013a; ACT Government 2017c). Rising CO_2 levels could potentially advantage trees and shrubs, facilitating further structural changes to grasslands, including an increased influx of woody weeds (Berry & Roderick 2005; Morgan *et al.*, 2007, cited in ACT Government 2017c) and predatory birds (Yeates, Osbourne & Gibbons, cited in Umwelt 2020). Ultimately, the threat posed by anthropogenic climate change to *S. plana* will likely vary heavily across its range and, while difficult to predict, in some instances it is likely to be severe (DSE 2009; Kutt *et al.*, 2015).

4.5 Genetic factors

The limited capacity of female *S. plana* to move across a landscape and colonise new areas limits the potential for gene flow between populations (Threatened Species Scientific Community, cited by Australian Government 2013). *S. plana* is notable for low population-level genetic diversity compared with other butterflies and moths (Clarke & O' Dwyer 2000). The high rate of non-random mating that has been observed in *S. plana* populations implies that individuals in smaller populations are likely to be very closely related, and potentially inbred (Clarke & O' Dwyer 2000). Geographically isolated populations of *S. plana* show increasing genetic differentiation. This could impose complications if future management requires translocation of individuals between sites (ACT Government 2020a). In a more immediate sense, the species has been proposed to be adapting to a diet of Chilean Needle Grass (*N. neesiana*), which could potentially lead to hardwired genetic changes. This could result in different local populations evolving to favour different food plants, and then becoming genetically isolated from one another (ACT Government 2017c; ACT Government 2020a). Adapting to feed upon *N. neesiana* could lead to a new reproductively-dominant strain of the species. This could potentially cause the loss of the genetics needed to maximise a life cycle at drier and sparser sites, with poor soil, and accordingly cause population declines in these areas (ACT Government 2017c).

The short adult lifespan, and very specific behaviour of this species, renders its reproduction vulnerable to altered conditions, which can potentially contribute to genetic drift and inbreeding depression (Clarke & O' Dwyer 2000). Changes to biomass levels and seasonal flight conditions can disrupt reproduction within a population, with the resulting low rates of recruitment then causing a loss of genetics (Kutt *et al.*, 2015). Similarly, male *S. plana* are attracted to broken yellow glass when it glints in the sunlight (Bainbridge & Longmore 2016) (see Appendix E3). This appears to register to the visual pattern recognition of the patrolling males as being the mating signal of a displaying female. As many as 20 moths per minute have been observed attempting to mate with glass shards, comprising a significant proportion of the total population counted over a daily survey. This probably both reduces the reproductive success of the affected males and increases their vulnerability to predators (Bainbridge & Longmore 2016). Mata *et al.* (2017) identified predation risk as the single biggest factor influencing adult survival at urban sites and suggest that its impacts on fragmented populations may be underappreciated and in need of more active management. As a given *S. plana* population declines, each successful mating event increases in importance, meaning that any effect that increases attrition amongst breeding adults can potentially endanger the entire population (Clarke & O' Dwyer 2000; Bainbridge & Longmore 2016). Census data gathered from a single *S. plana* population in the ACT implied that up to 99% of the potential fecundity of the population was unrealised due to unknown factors (Clarke & Dear 1998).

4.6 Introduced fauna

Currently, almost nothing is known about the impacts of invasive predators upon *S. plana* (Yeates, Osbourne & Gibbons, cited in Umwelt 2020). House Sparrows (*Passer domesticus*), Common Mynahs (*Acridotheres tristis*) and Common Starlings (*Sturnus vulgaris*) all congregate on perimeter structures around reserves during the flight season, in order to hunt the species (Bainbridge & Longmore 2016) (see Appendix E4). This increased predation appears to be an important factor in the continued survival of some populations in small, fragmented, urban sites, where the population is low enough that individual attrition has the potential to contribute to genetic bottlenecks (Clarke & O' Dwyer 2000; Bainbridge & Longmore 2016; Mata *et al.*, 2017). The impacts of invasive predators can be exacerbated by inappropriately planting trees and shrubs on sites occupied by *S. plana* (DEWHA 2009). Current research is investigating the potential impacts that predation by invasive insects plays upon *S. plana*, including the European Wasp (*Vespula germanica*) (Yeates, Osbourne & Gibbons, cited in Umwelt 2020).

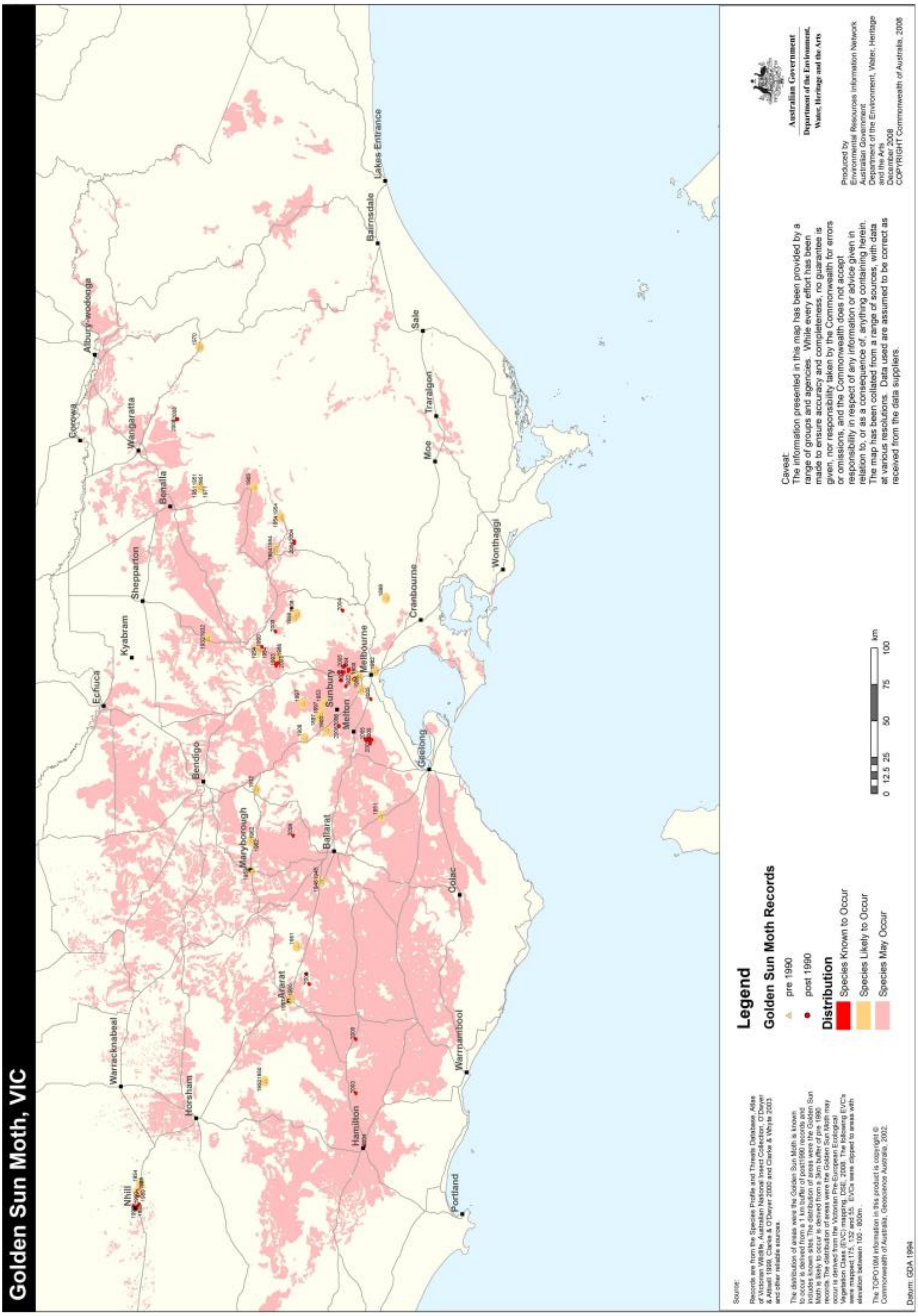
European Rabbits (*Oryctolagus cuniculus*) have contributed to both soil disturbance and an increased influx of weeds at several monitored sites (Richter *et al.*, 2009). High rabbit numbers can also contribute to over-grazing, structural changes to the soil and altered nutrient cycling. The erosion and degradation of site quality that results from rabbit activity are considered to pose moderate level risks to *S. plana* populations (DAWE 2021a). *O. cunciculus* is a deleterious species within Australian temperate grasslands, including high altitude systems in the ACT (ACT Government 2017d; Centre for Invasive Species Solutions 2021; Hartley *et al.*, 2021). Sheep grazing selects for a limited suite of grazing tolerant C3 grass species (Leigh *et al.*, 1989). Accordingly, rabbit activity tends to be less impactful to native grasses in such systems, with degradation concentrated to within 50 metres of warrens (Leigh *et al.*, 1989).

Chapter 5. *Synemon plana* populations in Victoria

Within Victoria, *S. plana* occupies temperate grassland and grassy woodland habitats across most of the state, including within the Wimmera and the northern Riverina (Richter *et al.*, 2009). More than 98% of Victoria’ s native grassy ecosystems have been radically altered by anthropogenic activities since early European settlement (DSEWPC 2011). After several new *S. plana* populations were discovered in Victoria, most subsequent survey work was concentrated within grasslands in north and west Melbourne and to the south of the Grampians (Brown *et al.*, 2012). Accordingly, Victoria’ s *S.plana* populations are largely represented by poorly documented rural populations and fragmented urban populations, with both categories facing ongoing threats from development (Biosis 2019b). Following increased survey effort, there is now known to be an abundance of *S. plana* populations in Victoria, with many occupying an extensive system of conservation reserves (SWIFFT 2022). Many of these populations nevertheless face uncertain futures, partly due to the potential impacts of anthropogenic climate change (Kutt *et al.*, 2015).

5.1 Background

A Victorian Government Threatened Species Assessment (cited in DAWE 2021a) projected that, under optimal conditions, the state's *S. plana* population comprised 13, 500 adults across any given flying season. Considering that the current lowest nationwide population estimate for *S. plana* is 10,000 individuals (DAWE 2021), the figure assigned to the Victorian population highlights *S. plana*' s comparative abundance within the state, as well as the general difficulty in assigning robust population numbers to this species. Large areas of Victoria lack adequate survey data for *S. plana* and consequently multiple populations likely remain undetected (Figures. 6-8) (OEH 2012, cited in DAWE 2021a). Victoria's *S. plana* population generally occurs at elevations of between 95 and 406 metres above sea level, with the species preferring sites with northerly facing slopes, of less than 4° (DSE 2004). *S. plana*'s ancient widespread occupancy of these landscapes has resulted in a high level of genetic diversity between populations (Clarke & Whyte 2003). The Temperate Grasslands of the Victorian Volcanic Plains (VVP) are the most important ecological community for *S. plana* in Australia.



Source:

Records are from the Species Profile and Threats Database, Atlas of Victorian Wildlife, Australian National Insect Collection, O'Dwyer & Attiwill 1999, Clarke & O'Dwyer 2000 and Clarke & Whyte 2003 and other reliable sources.

The distribution of areas were the Golden Sun Moth is known to occur is derived from a 1 km buffer of post1990 records and includes known sites.The distribution of areas were the Golden Sun Moth is likely to occur is derived from a 3km buffer of pre 1990 records.The distribution of areas were the Golden Sun Moth may occur is derived from the Victorian Pre-European Ecological Vegetation Class (EVC) mapping, DSE, 2008. The following EVC's were mapped;175, 132 and 55. EVCs were clipped to areas with elevation between 100 - 800m .

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Datum: GDA 1994

Caveat:
The information presented in this map has been provided by a range of groups and agencies. While every effort has been made to ensure accuracy and completeness, no guarantee is given, nor responsibility taken by the Commonwealth for errors or omissions, and the Commonwealth does not accept responsibility in respect of any information or advice given in relation to, or as a consequence of, anything containing herein. The map has been collated from a range of sources, with data at various resolutions. Data used are assumed to be correct as received from the data suppliers.

Produced by
Environmental Resources Information Network
Australian Government
Department of the Environment, Water, Heritage and the Arts
December 2008

Figure 6. A projected distribution of the range of *S. plana* populations and habitat within Victoria, as of 2008 (DEWHA).

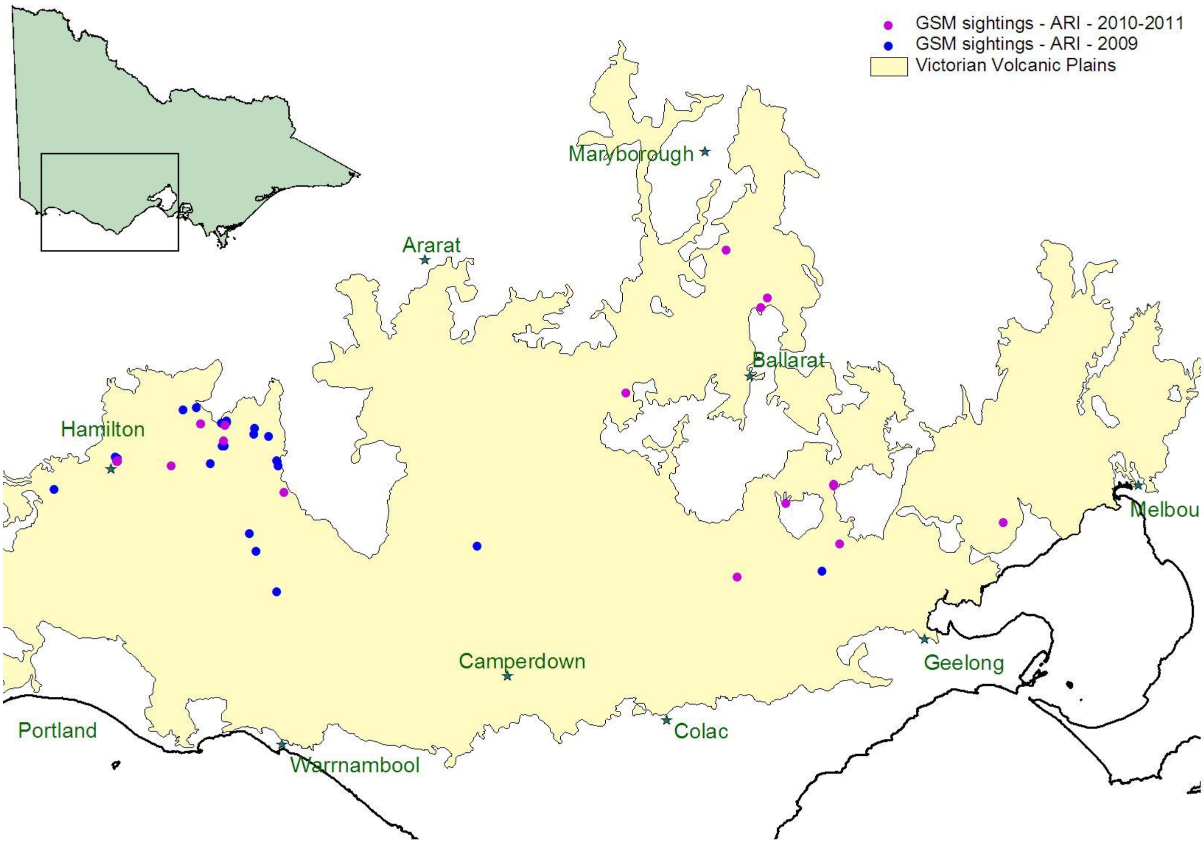

Figure 7. Recent additions to known *S. plana* populations within Victoria, following statewide surveys in the years (2009-11), These surveys were limited in scope by site access and suboptimal conditions. Reproduced with the kind permission of Arn Tolsma and the Arthur Rylah Institute (2022).

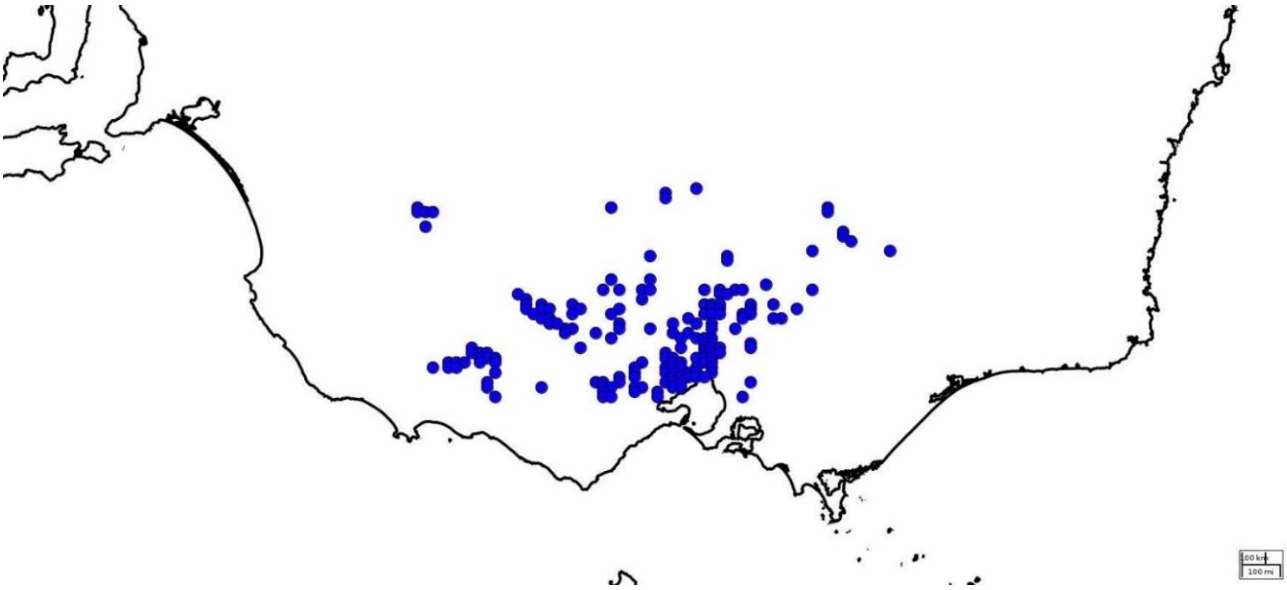

Figure 8. All current records of *S. plana* In Victoria, Including additions to ARI surveys (2009-2011, Figure 6) (Victorian Biodiversity Atlas, 2022).

Large expanses of these systems are naturally almost treeless and are dominated by *Themeda triandra* in association with a complex of other native grasses and a high diversity of native forbs (DSWEPC 2011; Sinclair & Atchison 2012; DAWE 2021a). While the primary habitat for the species in Victoria is temperate grassland, Victorian populations of *S. plana* are also associated with VVP Grassy Eucalypt Woodland as secondary habitat. These woodland systems are typically dominated by River Redgum (*Eucalyptus camaldulensis*), however in drier areas, such as within rain shadows, either Grey Box (*E. microcarpa*) or Yellow Box (*E. melliodora*) can be locally abundant (DSE 2009). Both ecosystems support a broad community of other threatened taxa including the Striped Legless Lizard (*Delma impar*), the Matted Flax Lily (*Dianella amoena*) and Clover Glycine (*Glycine latrobeana*). The C3-dominated grassland systems that are favoured by *S. plana* in Victoria are generally considered to be of lower conservation value than other VVP grassland habitats, because they support a reduced suite of native taxa (Sinclair *et al.*, 2014).

Historic records show that *S. plana* was once prevalent in central Victoria, and occupied grassland habitats extending from Bordertown in South Australia, across the Wimmera Plain and the northern Goldfields, and extending into grasslands around Wangaratta and Bright (DSE 2004; Richer *et al.*, 2009). These populations once sprawled across the Central Victorian Uplands (Kutt *et al.*, 2015), and were once recorded as far east as Mansfield and Eildon (DSE 2004). Today, most documented populations occur in the west of the state, on the VVP, as well as on the western and northern edges of Melbourne (Brown *et al.*, 2012; Richter *et al.*, 2013b). During the early twentieth century, agricultural development of the open landscapes and fertile soils of the VVP precipitated one of the most intensive landscape-scale modification events in Australian history. This development resulted in more than 95% of the previously dominant VVP Grassland and Grassy Eucalypt Woodland communities being removed through clearing, cropping and grazing (Stuwe 1986 & McDougall *et al.*, 1994, *cited in* Morgan 1998; DSEWPC 2011). As a result, VVP Grasslands and Grassy Woodlands are now both listed as Critically Endangered Ecological Communities (DSEWPC 2011). Most of the surviving remnants are less than 10 hectares in size, and fewer than a quarter of these represent high quality habitat in an intact condition (Morgan 1998; DSEWPC 2011). The best quality VVP remnants generally occur in cemeteries, railway lines and road verges, where they have escaped agricultural conversion (Morgan 1998; DSEWPC 2011; Sinclair *et al.*, 2014). The remainder of these communities are mostly degraded as a result of weed and pest invasion, which has caused a pronounced loss of biodiversity (Morgan 1998; DSEWPC 2011). Consequentially, these grassland remnants require concerted ecological remediation to restore their function (Morgan 1998; DSEWPC 2011). Historically, VVP grassland and grassy woodland habitats have been poorly represented within conservation reserves. Many VVP sites have faced uncertain futures on private land, with the ever-present potential for future development or disturbance. A notable minority have, however, benefited from forward thinking land management that sought to retain the biodiversity of these sites (Parliament of Victoria 2021). Within Melbourne, urban sprawl has similarly reduced the biodiversity of the city’s footprint to a fraction of its pre-colonial extent and caused the destruction of 96% of the city’s VVP grassland habitat (DSE 2009). Victoria’s first reported *S. plana* population extinctions occurred in 1938 (Atlas of Victorian Wildlife, *cited in* Dear 1997). *S. plana* numbers began to decline throughout its national range in the latter half of the twentieth century and the species was presumed extinct at most known sites in Victoria over subsequent decades (Edwards 1993, *cited in ACT* Government 2017a).

5.2 Rediscovery across Victoria

In 1992, a population of *S. plana* was discovered at Mt Piper, in the Central Uplands, which was, at that time, among the largest populations known in Australia (Britton *et al.* 1995; Clarke & O'Dwyer 2000) (Appendix E5). Two additional *S. plana* populations were discovered over the following three years: the first nearby at Tallarook, and the other at Salisbury Bushland Reserve in the Wimmera (Clarke & O'Dwyer 2000). By 1999, the species was known from five locations in Victoria (O'Dwyer & Attiwill 1999). A genetic review of *S. plana* populations in Victoria established that the currently known Victorian populations were representative of a separate genetic haplotype to the more widely known populations in the Australian Capital Territory (ACT) and New South Wales (NSW). Secondly, the genetic review identified high rates of genetic divergence based upon the geographic distance between the known Victorian populations (Clarke & O'Dwyer 2000). Clarke and Whyte (2003) identified a particularly high level of distinction among specimens from Nhill, in the Wimmera. A small percentage of female *S. plana* at Wimmera sites comprise a distinctive local colour morph, in which the hind wings are creamy white, rather than gold (Douglas 2004). Similarly, the males of this area adopt an idiosyncratic camouflage behaviour, by resting with their wings in an arched, upright position, like that of a butterfly, which allows them to blend in against a background of dry grasses and seedheads (Douglas 2004). The authors of these studies proposed that Victoria's *S. plana* were suitably distinct from northern populations of the species to either constitute a distinct race, or potentially even a separate subspecies. The authors further asserted that Victorian populations required specific, targeted conservation measures to reflect this (Clarke & O'Dwyer 2000; Clarke & Whyte 2003). The genetic distinctiveness of Victoria's *S. plana* stems from deep-time evolutionary processes rather than from the contemporary fragmentation processes, that have arisen from European land use. Clarke, Whyte and O'Dwyer's work highlighted a need to protect Victoria's geographically isolated populations, in order to conserve the broader genetics and resilience of the species (Clarke & O'Dwyer 2000; Clarke & Whyte 2003).

S. plana was discovered at two sites in north-west Melbourne in 2003, by the Merri Creek Management Committee (Enderby & Koehler 2006), who secured a grant to convene an informal working group to survey for *S. plana* across the Merri Valley (Bainbridge & North 2007). A particularly large population of flying males was recorded at Craigieburn Grassland, establishing this reserve as the largest *S. plana* site in the state (Gibson & New 2007). These discoveries prompted more detailed surveying of grassland remnants around Epping, with four further populations discovered over the following two years in *T. triandra*-dominated grassland remnants (Enderby & Koehler 2006). *S. plana* was subsequently rediscovered at several more sites in Craigieburn, Epping and Deer Park, highlighting the potential for *S. plana* populations to remain undetected within any area of suitable habitat across its range (Enderby & Koehler 2006; New *et al.*, 2007). Many of these new populations were isolated and vulnerable to development pressure and habitat destruction (Braby & Dunford 2006; New *et al.*, 2007). A large population in Epping was destroyed in 2008 to construct a wholesale fruit and vegetable market (Bainbridge & Longmore 2016; State of Victoria 2020). In total, more than 27 new *S. plana* populations were discovered in Melbourne between 2003–2008, establishing that, although patchily distributed, the species was widespread around the northern and western outskirts of Melbourne (Gilmore *et al.*, 2008) (see Appendix E6). The Arthur Rylah Institute (ARI) surveyed the broader VVP during the 2009/2010 and 2010/2011 flight seasons. The ARI’s surveys discovered many new populations to the south-west of the Grampians; around Dunkeld and Hamilton, as well as additional scattered populations north of Ballarat and near Meredith (Brown *et al.*, 2012, *Figure* 6.). These surveys were disproportionately focused on west Victorian grasslands and survey attention was particularly sparse across the northern half of the state. Suitable *S. plana* habitat in the northern Goldfields disproportionately occurs on private property, which created site access issues and limited the reach of these surveys. 2010 surveying was further compounded by La Niña rains, which led to very low numbers of flying males recorded and a disproportionate representation of *S. plana*’s abundance across this area of the state (Brown *et al.*, 2012). Although work continues to catalogue and protect *S. plana* across Victoria, this effort has struggled to map its current state-wide distribution, which impedes efforts to conserve the species.

5.3 Current and future status

A projected 3,278 ha of native grassland is expected to have been destroyed in Melbourne by 2040, due to the revised Urban Growth Boundary (DSE 2009), with its associated residential and industrial expansion presenting the single most destructive process to VVP remnants in the city (Biosis 2019a). This development pressure has continued to drive the destruction of *S. plana* populations across Melbourne (New *et al.*, 2007). In recent times this has included the removal of several small populations near Craigieburn Grassland due to the development of an old quarry (Bainbridge & Longmore 2015), the removal of a population at Ravenhall by the Department of Justice to construct a prison facility (Biosis 2012) and the destruction of a larger population in Sunshine West by Australia Post, in order to build an industrial warehouse facility (Biosis 2019a). Developers are legally required, under state law, to compensate for the habitat destruction of significant populations of *S. plana* (New 2015). Such habitat is defined as areas where five or more male moths have been detected across a flight season, through formal survey (New 2015). The Victorian Government has committed to protect 80% of the remaining high-quality *S. plana* habitat in Melbourne, aiming to secure 8510 ha of grassland habitat through the 15,000 ha Western Grassland Reserve project (DEPI 2013) along with multiple small grassland reserves that each retain *S. plana* populations (Bainbridge & Longmore 2016). The creation of these new reserves is considered sufficient to offset the destruction of further *S. plana* populations within the Urban Growth Boundary (DEPI 2013). As of 2017, just 13.6% (1671 ha) of the highest priority *S. plana* habitat in Melbourne had been acquired (DELWP 2018). In 2020, the conservation status of Victoria's *S. plana* population was revised from Critically Endangered to Vulnerable, due to the high numbers of newly discovered populations, as part of the Conservation Status Assessment Project (DELWP 2021; SWIFFT 2022).

Nevertheless, the Victorian National Parks Association (2021, *cited in* Parliament of Victoria 2021) attest that they consider the VVP grasslands west of Melbourne to be underrepresented within the state's national parks and conservation reserve system and highlighted the government's failure to acquire the majority of the land designated to the Western Grassland Reserve (Victorian National Parks Association 2020). Parks Victoria and other land management authorities have suffered pronounced funding cuts. These cuts limit their capacity to pay sufficiently experienced land management contractors to maintain many of these sites with the finesse that these complex systems require (F. Sutton, Ecological Consultants Association of Victoria (ECA), *cited in* Parliament of Victoria 2021). In rural Victoria, farmers have strong economic incentives to intensify the productivity of their land through pasture enrichment and cropping (Dorrough *et al*, 2007). Accordingly, there is a pressing need for targeted government biodiversity protection in rural Victoria (Dorrough *et al.*, 2007). In the interim, *S. plana* offsets have been established in regional areas (Biosis 2019a), seeking to address the absence of a government strategy to protect *S. plana* populations in regional Victoria and to provide broader-scale population resilience to the species (Biosis 2019b). While *S. plana* is now well-protected within Melbourne, its general population health across regional Victoria is harder to gauge, due to a paucity of survey data and a lack of targeted policy to protect the species across these areas.

In contrast to both the western Victorian *S. plana* sites, and the sites that the species occupies in Melbourne, comparatively little is known about the northern Victorian distribution of *S. plana*. This lack of information results from a general paucity of past survey attention in this part of the state (Brown *et al.*, 2012; T. Wills 2022, *pers. coms.*). Ninety percent (90%) of the remaining suitable habitat in the broader Goldfields Bioregion occurs on private land (North Central Catchment Management Authority, 2005, *cited in* Hepburn Shire Council 2018). Survey attempts to identify remnant *S. plana* populations in central Victoria remain ongoing, with the known populations typically confined to roadsides (Hepburn Shire Council 2018; Macedon Shire Council 2018). Some *S. plana* populations have survived on farms in this region, because their proximity to basalt rock escarpments has provided them protection from ploughing (Biosis 2018b). In recent years, farmers have increasingly begun to deploy industrial machinery to remove rocks and initiate ploughing of such landscapes (Biosis 2018b). The lack of development pressure across the northern plains has led to the retention of many grasslands and grassy woodlands that contain suitable *S. plana* habitat. These areas have not yet received the intensive survey attention that has been necessary within Melbourne (M. Venosta, Biosis, 2022, *pers coms.*). Three large *S. plana* populations were identified in Ganawarra Shire in 2008 (Victorian Biodiversity Atlas 2022, *Figure* 7.), the species was discovered at HP Barr Reserve in Wangaratta in 2021 (Rural City of Wangaratta) and a new population was discovered at Ninyeunook in 2009, establishing a new north-western edge for *S. plana* habitat in Victoria (Trust for Nature, *cited in* Biosis 2013). Additional populations likely await discovery in the surrounding landscape (SWIFFT 2022). In 2010, several sites in the Wimmera, at the far north-west of the species' current range, were the sites for a project to establish new populations through translocation, in order to further bolster the presence and resilience of *S. plana*. across the region (SWIFFT 2022).

Some contemporary reviews have extended potential *S. plana* habitat in Victoria to include areas of grassland in the east of the state (*Figures,* 3 & 5, DAWE 2021a; DEWHA 2008). Other than some old records that are comparatively close to Melbourne, *S. plana* has never been observed in eastern Victoria. Some of this proposed habitat incorporates Natural Temperate Grasslands of the South-eastern Highlands, which is the characteristic ecosystem that *S. plana* occupies in the ACT and NSW (DEE 2016). The lack of targeted surveys for *S. plana* in these landscapes, combined with the cryptic nature of *S. plana*, and the general scarcity of suitable habitat in the broader landscape, may have led to the species simply being missed within eastern Victorian sites (J. Morgan, 2022. *pers. coms.*). Inverted tree line grass pockets occur, for example, on the slopes of Mt Buffalo (New 2019). These grassland communities occupy frost-hollows (New 2019), such as are associated with *S. plana* habitat in the ACT (Dear 1997). Climatic factors are critically important to the long-term survival of this species, both in geographic extremes, and across its broader distribution.

Predicting the future survival of *S. plana* populations in Victoria, under a changing climate, is fraught with difficulty, and likely to vary heavily from site to site (DSE 2009; Kutt *et al.*, 2015). Greater Melbourne is projected to become both hotter and drier over upcoming decades, especially during the summer (DSE 2008). By 2030, annual temperatures are expected to be 0.8°C higher than 1990 levels, with an accompanied increase in days with temperatures exceeding 30°C (DSE 2009). Annual rainfall, especially during the spring, is projected to decrease by 4-7 % (DSE 2009). If emissions continue to follow the higher end of projected estimates, Melbourne, by 2070, may have an annual rainfall cycle resembling that of Seymour, and experience annual temperatures similar to Echuca (DSE 2009). Isolated urban sites will face increased environmental stresses, including weed and pest invasions, increased frequency of grassfires and changes to annual soil moisture content, with uncertain consequences for *S. plana* (Kutt *et al.* 2015; T. Wills, *pers coms.* 2022). Victorian *S. plana* grasslands that occur under rain shadows, such as those around the You Yangs, are likely to be severely impacted by reduced annual rainfall, which may facilitate broad declines in C3 food grasses needed by *S. plana*, and eventual conversion to chenopod shrublands, unsuitable for the species (DSE 2009; SWIFFT 2022). Parks Victoria identify climate change as a major factor limiting the effectiveness of their conservation management, under currently available budgets (M. Norman, *cited in* Parliament of Victoria 2021). An inquiry into ecosystem declines across Victoria, recommended that the Victorian Government increase Parks Victoria's funding to one percent of the Gross State Product, in order to allow the organisation to tackle the emerging threats of anthropogenic climate change (Parliament of Victoria 2021).

Chapter 6. Practical Management of *Synemon plana*

Resources for *S. plana* conservation should be allocated based upon the perceived long-term viability and potential genetic significance of populations, which will vary from site to site (DAWE 2021b). Managing a site to support *S. plana*, requires the maintenance of an appropriately structured grass sward, dominated by C3 food grass species (DEWHA 2009). *S. plana* occurs within several categories of Victorian grassland but has particularly arisen to prominence as a flagship species for Threatened Victorian Volcanic Plains landscapes (DAWE 2021a; New 2011). However, the C3 grass dominance at many of the sites *S. plana* occupies in Victoria does not reflect pre-European settlement conditions (Sinclair *et al.*, 2014). Moreover, *S. plana*'s biomass requirements are potentially undesirable for other threatened grassland taxa with different habitat preferences (e.g., Striped Legless Lizard (*Delmar impar*) (ACT Government 2017c). Maintaining *S. plana* habitat involves intensive biomass reduction approaches that emulate broader scale ecological processes such as naturally occurring wildfires and kangaroo grazing (ACT Government 2017c). Current management options include stock grazing, ecological burning and slashing with biomass removal (ACT Government 2017c). A robust and quantitative understanding of the floral community of the site, and the state of its biomass, forms a key tenet for such management activities (Kutt *et al.*, 2016).

6.1 Overview

The paucity of knowledge on the lifecycle and the physical distribution of *S. plana* presents a hurdle for the management of existing populations. These knowledge shortfalls have impeded attempts to produce a reliable and consistent set of approaches that best provide for the ecological needs of *S. plana* (Gibson and New 2007; Richter 2013a). Any potential grassland or grassy woodland within the distribution of *S. plana* could potentially support the species, provided that the site either retains native grasses or has been invaded by *N. neesiana* (Enderby & Koehler 2006). The protection of large, intact sites of high-quality habitat is currently the primary emphasis for conservation of this species in Victoria (DEPI 2013). *S. plana*'s lowered threat status has reduced the relative value that populations in small and degraded areas hold for the overall survival of the species (DAWE 2021b). However, invertebrates have historically been understudied and underrepresented within conservation management (Eisenhauer *et al.*, 2019; New 2019), so the assigned population numbers currently providing the foundation for the conservation status of *S. plana* provide less robust indicators than would be the case for a vertebrate species in an analogous scenario (Sands 2018; G. Robertson, Friends of Grasslands, 2022, *pers coms*). Indeed, gaps in current data, combined with a lack of consolidated field knowledge prevent the identification of clear population trends for the majority of Victoria's threatened taxa (J. Morgan, B. Wintle, M. O'Shea, C. Nitschke 2021, *cited in* Parliament of Victoria 2021). Accordingly, a precautionary approach is to provide protection for *S. plana* within existing management programmes wherever feasible, especially due to the large knowledge gaps that remain with regards to the *S. plana* lifecycle.

Smaller sites can provide protection for genetically distinct populations of *S. plana* in heavily fragmented areas (Clarke & Whyte 2003) or can serve more generally as insurance populations, should a catastrophic event that eliminates an entire *S. plana* population occur at a larger protected area (DEPI 2013). However, smaller and more fragmented *S. plana* sites generally require more intensive ongoing management (DEPI 2013; Richter *et al.*, 2013a). These small sites incur a slew of edge effects, particularly in urban areas, which include weed incursions, changes to soil chemistry, increased levels of avian predation, exposure to dumped rubbish and other anthropogenic factors, all of which require additional resources to mitigate (Bainbridge & Longmore 2016; Mata *et al.*, 2017). General biomass reduction measures, such as ecological burns, are more dangerous in residential and industrial areas and can lead to opposition from adjacent stakeholders (Mata *et al.*, 2017).

Offsetting the destruction of low-priority populations by protecting and bolstering populations in remote areas of Victoria, rests on the presumption that the biology of the species, and the techniques available to maintain its habitat, are sufficiently well understood that site management efforts can produce such outcomes in the first place (see New 2012). Initial *S. plana* offsetting in Victoria struggled against pressure from developers to sign-off degraded and unsuitable areas as legal *S. plana* offsets, which was exacerbated by uncertainties about the suitability of *N. neesiana* as a food plant (T. New 2022, *pers. coms.*). Offset management plans (OMP) for *S. plana* increasingly require that management actions produce measurable benefits to a population, which managers must demonstrate through recording higher numbers of flying adults in population counts across subsequent seasons (Various site owners interviewed, *pers. coms.* 2022). Concerns remain about Threatened Species Offsetting in Victorian native grassland environments, particularly regarding the quality of the new sites presented as representative habitat for those lost to development (Victorian National Parks Association 2020; J. Crook, Grassy Plains Network, *cited in* Parliament of Victoria, 2021).

Roughly 22% (3.1 million Ha) of the agricultural land of south-east Australia is unimproved "native" pasture (Mitchell *et al.*, 2019) with appropriate soil conditions for *S. plana* conservation. Any area utilised for *S. plana* conservation must retain appropriate soil chemistry for both *S. plana* larvae and for the native grasses that they feed upon (DES 2004). Fertilisers are potentially toxic to *S. plana* larvae, and they facilitate weedy grasses outcompeting their food plants (O' Dwyer & Attiwill 1999; Douglas 2004). On a farm offset, this means ensuring that any fertilisers applied to soils do not increase the nitrate levels above those that would naturally occur in native grasslands, but this also requires that managers are attentive to prevent fertilisers from leaching into the soil from adjacent blocks or from being blown on the wind and dispersed into the offset (DSE 2004). This generally means that no soil fertilization is possible within an area set aside for *S. plana* conservation. Managing appropriate biomass on an *S. plana* site is essential in order to provide the species with the food plants and inter-tussock spacing that it requires (DEWHA 2009). All available approaches to achieve this come with their own associated complications that must be carefully weighed against the desired outcome. As a rule of thumb, introducing intensive, novel management changes to high quality areas presents a high risk of upsetting existing ecological processes and then causing the site to decline in quality (Lunt 1991; Lunt *et al.*, 1997). Long unburnt areas are difficult to return to their desired state, simply by introducing ecological burns, because they will burn unevenly and at unsuitably high temperatures (Morgan & Salmon 2019). Sheep tend to avoid grazing such patches as well, because they selectively browse upon green grass blades and fresh tillers over dead biomass (N. Oddie 2022, *pers coms*).

Stocking numbers, and the duration of grazing times are important considerations on an *S. plana* offset (Greenwood & McKenzie 2001; Mata *et al.*, 2017) given the ease with which stock grazing can upset the ecological processes of a native grassland and convert it to dominance by exotic annuals (Neave & Tanton 1989; Dorrough *et al.*, 2004; Lunt *et al.*, 2007). Crash grazing in offsets that are simultaneously used as stock grazing habitat must be carefully timed in order to remove the target weeds before depleting the food resources of the stock and impacting desirable flora (R. Bellchambers 2022, landowner and grazier, *pers coms.*). Cattle are particularly likely to compact soil, which causes the site to degrade in quality, affecting resident *S. plana* (Greenwood & McKenzie 2001; Mata *et al.*, 2017). Despite this, high *S. plana* populations have nevertheless persisted for decades under cattle and horse grazing, at certain sites (Bainbridge & Longmore 2015). Without careful, proactive management, even relatively light sheep grazing can cause soil compaction with similar impacts to larger stock, particularly in dry conditions (Greenwood & McKenzie 2001; Mata *et al.*, 2017). The intensity and duration of grazing, that is suitable at a site will depend upon its soil type and location, with some regions being more prone to erosion and pasture depletion from heavy stocking than others (N. Oddie, landowner and grazier, 2022, *pers coms.*). Stock grazing is particularly risky in the wet season habitat refugia on which *S. plana* depends, where it has higher potential to impact the population (Kutt *et al.*, 2016; T. Wills, 2022, *pers coms.*). Installing internal fences to restrict stock movements can impose associated compounding impacts upon grassland systems, such as creating concentrated patches of increased disturbance and soil nitrate loading, through the formation of additional sheep camps (Niu *et al.*, 2008; Maheswaran *et al.*, 2022; N. Oddie*, pers coms*, 2022), as well as providing perching habitat for predators during the *S. plana* flight season (Bainbridge & Longmore 2016).

Following careful planning and preparation, an effectively deployed ecological burn not only reduces the biomass on a site, but can also stop weed seeds from accumulating in the topsoil, and reduce soil nutrient levels, thus aiding its transition back towards native species dominance (Douglas 2004; Biosis 2019b; Biosis, *cited in* DEWHA 2009). However, *S. plana*' s capacity to withstand fire events remains contentious and lacking in robust data. Records of adult moths emerging after an ecological burn can potentially be confounded by *S. plana* recolonising the area from adjacent patches and may underrepresent their historic rate of mortality during burns (Edwards 1994, *cited in* ACT Government 2017). Additionally, there are broad accounts of large populations of *S. plana* surviving on unburnt sites across multiple decades (Edwards 1994, *cited in* ACT Government 2017; T. Fitzgerald, Parks Victoria, 2022, *pers. coms.*), although, invariably, such areas have extended histories of stock grazing (e.g., ;Dear 1997; Bainbridge & North 2007). Summer burning provides a useful weed control measure in grassland restoration, because it depletes the surface soil of moisture, thus advantaging deeper rooted native plants over shallower-rooted weed species (R. Bellchambers 2022, *pers coms.*). However, a prudent approach requires avoiding burning during *S. plana* pupation (September-November), adult emergence and egg-laying (October-January), in order to accommodate for Edwards' initial concerns, in the current absence of robust scientific data (Biosis, *cited in* DEWHA 2009). These restrictions form legally binding criteria for many current *S. plana* offset plans in Victoria, and often OMPs will require site managers to attain written permission from the Department of Environment, Land, Water and Planning (DELWP) before attempting a burn (various interviewed landowners 2022, *pers. coms.*). Data paucity on the early lifecycle of *S. plana* does not allow for accurate predictions of the impacts of fire either upon eggs within the soil surface or hatchlings within tussock bases. *S. plana* hatchlings may be present above the soil throughout the year (Richter *et al.*, 2013a).

The species composition of a grassland plays an important role in how a given site responds to an ecological burn, which will directly affect how suitable this approach is for *S. plana* management (Lunt *et al.*, 2012; Morgan & Salmon 2019). There is some evidence that high standing fuel loads within *Austrostipa*-dominated grasslands can cause higher fire intensity than typically occurs within the better-studied *T. triandra*-dominated grasslands on which most working industry knowledge of ecological burns has been developed, although such differences appear minor (Morgan & Salmon 2019). The lack of long-term data on how C3 Grasslands, particularly those dominated by *Rytidosperma*, respond to fire, encourage particular care when applying burns to such landscapes, due to the potential for unanticipated outcomes to desirable species (Sinclair *et al.*, 2014; Morgan & Salmon 2019). The response of different grass species to fire cannot be readily predicted by the photosynthetic pathways engaged, but maps more readily to evolutionary history of the grass species and the environment to which it is adapted (Ripley *et al.*, 2015). Anecdotally, TREC Land Services (*pers obs.*, 2022) have frequently observed that the dense structure of *T. triandra*-dominated systems, retains structural moisture low to the ground, producing an even smouldering burn. In contrast, the *Rytidosperma* and *Austrostipa* species typical to a VVP grassland produce a drier and less dense structure, with higher levels of elevated dry biomass that causes tussocks to burn more rapidly (TREC Land Services 2022, *pers obs.*). High tussock mortality has previously been observed within disturbed *Austrostipa*-dominated grasslands following abrupt exposure to fire (Sinclair *et al.*, 2019).

6.2 Costs and Benefits of *Synemon plana* Conservation

S. plana is one of the most iconic moths in south-eastern Australia, and has become an emblematic species for Australian conservation, particularly for that of temperate native grasslands (Richter *et al.*, 2009; Mata *et al.*, 2017). *S. plana* is widely promoted as an umbrella species for grassland conservation, and even after its Threat Status in Victoria was downgraded to Vulnerable, its continued prominence leads to increased funding and protection for both the Critically Endangered Natural Temperate Grassland and the Grassy Eucalypt Woodland communities of the Victorian Volcanic Plains (VVP) that comprise much of its habitat. Protecting *S. plana* habitat provides grassland sites with an additional buffer of protection from development, which directly reduces the rate at which they are cleared (DAWE 2021a). Conservation management activities for *S. plana* provide flow-on benefits to a broad community of grassland and grassy woodland species, including preserving the biodiversity of native plant communities at these sites, retaining their natural ecological processes (DSE 2004) and providing protection to lesser-known invertebrates that lack their own management plans (New 1997). The process of studying and conserving this flagship moth provides an avenue for personnel to increase their expertise in insect conservation, which has flow-on benefits for other threatened species and broader conservation projects (DSE 2004). These effects are magnified at a site if combined with the promotion of other flagship species. At Mt Piper, the presence of *S. plana*, in conjunction with two threatened Ant Blue Butterfly species (Large Ant Blue (*Acrodipsas brisbanensis*), Small Ant Blue (*A. myrmecophila*)), has led to increased attention to the broader invertebrate community and more focus upon some of its lesser-known members (New 1997; B. Tomkins 2022, FoMP 2022, *pers coms.*). This systemic and detailed understanding of an ecological area's invertebrate community is useful as a general site quality indicator and it aids understanding of the broader habits and distributions of each species involved (Kutt *et al.*, 2016). Some of the oldest *S. plana* grassland reserves in Canberra are now listed heritage sites, due to their decades-long histories of scientific investigation (Umwelt 2020).

Grassland conservation is, at the crux, controversial, and subject to broad opposition from developers on economic grounds (New *et al.*, 2007). The relative economic value of the land occupied by *S. plana* is generally highest in urban areas, such as within Melbourne's western and northern urban growth boundary (New 2014, *cited in* New 2016). Such areas are intensively sought after for housing, industry and the creation of entire new suburbs (New 2016). Accordingly, *S. plana* has been described as the "meat in a complex sandwich between developers and conservationists" (New 2012; New 2018).

The general public can also oppose ecological burns or the timing of other biomass measures on nearby grassland reserves, due to perceived safety concerns, particularly in urban areas (Mata *et al.*, 2017). Landowners that do agree to the creation of an *S. plana* offset on their property are required to commit to an Offset Management Plan (OMP), which will heavily restrict the land management practices that they can perform on the designated area of the site and can formalise very specific ongoing targets that may span decades. An OMP is a legally binding document that the site owner must then abide by for the specified time period, come what may, or face non-compliance regulation and its consequences (various interviewed *S. plana* offset owners, 2022).

Managing a grassland for *S. plana*, with the aim to create the desired inter-tussock spacing of bare soil patches, is broadly beneficial to a range of other taxa, and the resulting habitat structure is seen as the default template state for a healthy temperate native grassland ecosystem (DEE 2011). However, the specific needs of more sensitive biodiversity often need to be weighed against this optimised approach (ACT Government 2017c; J. Morgan 2022*, pers coms.*). In general, preserving and enhancing these landscapes reduces soil erosion, limits the spread of weeds, stores carbon, benefits pollinators and aids the water retention of soils. These factors are in turn beneficial to human wellbeing and assist the agricultural productivity of nearby areas (DEE 2016; New 2019). If *T. triandra* is left to form a dense sward of mature tussocks with no inter-tussock spaces, it will choke out a broad range of native flora (Morgan & Lunt 1999). Depending upon the history of the site, regularly burnt areas can retain a suite of flora that depend upon maintenance of this low biomass state for recruitment (Morgan 1998), including species of legume with seeds that require fire to germinate (Bradstock & Auld 1995). Threatened taxa associated with this habitat include the Striped Legless Lizard (*Delma impar*) and the Matted Flax-lily (*Dianella amoena*), as well as the Critically Endangered Spiny Rice-flower (*Pimelea spinescens*) (DEE 2011) and Button Wrinklewort (*Rutidosis leptorrhynchoides*) (ACT Government 2017c). On the Wimmera Plain, *S. plana* grasslands provide habitat for the Endangered Pale Sun Moth (*S. selene*), including its Critically Endangered Nhill and Narrow-winged morphs (Douglas 2004). The Critically Endangered Grassland Earless Dragon (*Tympanocryptis pinguicolla*) is also associated with *S. plana* habitat (ACT Government 2017c; Robertson & Evans 2009).

However, the low biomass and high disturbance levels needed to attain the structure and species composition favoured by *S. plana*, are deleterious to a range of other species that occupy these landscapes (ACT Government 2017c; J. Morgan 2022*, pers coms.*). These processes can lead to a reduction in birds, reptiles and insects (Antos & Williams 2015; Sands 2018), as well as a reduction in floral species that are either grazing-sensitive (Lunt *et al.*, 2007) or have specific fire requirements unsuited to an *S. plana*-focused management regime (Bradstock & Auld 1995). *Delmar impar*, for example, favours a higher level of biomass than *S. plana*, and typically occurs on sites with moderately tall tussocks and forbs, and is thus potentially impacted by management practices aimed specifically at *S. plana* conservation, such as routine mowing or slashing (Howland *et al.*, 2004). *D. impar*' s long-term tolerance to a uniformly structured habitat of short grass is not well understood, and its conservation needs will play an important factor at any site that this species remains present (ACT Government 2017c). If smaller reptiles decline across a grassy ecosystem, correspondingly, this deprives birds and larger reptiles of food resources, which can lead to resulting population declines at sites supporting native predators of conservation value (Howland *et al.*, 2004). In a Victorian context, these threatened predators, where present, tend to be more associated with grassy woodlands and include Little Eagle (*Hieraaetus morphnoides*) (Scientific Advisory Committee 2020), Square-tailed Kite (*Lophoictinia isura*) (SWIFFT 2022) and Murray/Darling Carpet Python (*Morelia spilota metcalfei*) (Wilson & Swan 2017). In general, whenever a site is exclusively managed for *S. plana* conservation, this will lead to associated impacts on other taxa, and these processes will be especially pronounced within small, isolated sites. For larger areas, depending on the site' s management goals, and the specific taxa present, it is sometimes necessary to limit *S. plana*-focused conservation to small patches, while allowing adjacent areas to retain heterogeneous levels of biomass within a broader mosaic, to support a diverse ecological community (ACT Government 2017c).

For similar reasons, managing secondary grasslands and grassy woodlands as primary *S. plana* reserves comes with a slew of associated problems. The grassy clearings that *S. plana* occupy in such landscapes cannot support the species if the surrounding trees and shrubs are allowed to regenerate.

Accordingly, retaining an open sward grassland state in such areas requires intensive ongoing disturbance regimes, which come at the expense of the mammals, reptiles, invertebrates and some of the flora that normally occupy the area's woodland habitat (ACT Government 2017c). Many threatened species of flora do not leave persistent soil seed banks, so rarer species can rapidly disappear from a system under such intensive changes to disturbance regimes (Morgan 1998; Lunt *et al.*, 2007). Reductions to understory vegetation can also increase exposure of vulnerable faunal species to invasive predators (e.g., Red Fox (*Vulpes vulpes*)) (Saunders *et al.*, 2010). As more *S. plana* populations have been discovered and an increasing number of these have received permanent protection in large conservation reserves, these kinds of concerns have become increasingly prevalent in the contemporary management of Victorian grassland sites that support *S. plana* (E. Stone, HCC 2022, *pers coms.*).

N. neesiana is utilised by *S. plana*, but this grass species is a noxious Weed of National Significance, that severely impacts the biodiversity of native grasslands (Richter *et al.*, 2013a), which land managers are normally mandated to control (DAWE 2021a). The specific needs and priorities of a given grassland reserve will determine whether the benefits that *N. neesiana* provides to *S. plana* should outweigh its broader impacts to other taxa. High priority *S. plana* populations that occupy small, fragmented sites, or populations that occupy heavily degraded areas with few remaining native grasses, both present potential situations in which retaining *N. neesiana* is of greater benefit to *S. plana* than its removal would be for the broader ecological community (Richter *et al.*, 2013a; Mata *et al.*, 2017). Incursions of *N. neesiana* can therefore improve a degraded area's suitability as *S. plana* habitat, which may also prove to be true for its sister species *N. trichotoma* (also a listed Weed of National Significance), should *N. trichotoma* indeed turn out to be a food plant for *S. plana* (DAWE 2021a).

6.3 Current best practice approaches in Victoria

S. plana can potentially occur at any grassland or grassy woodland within its distribution, that either retains native grasses or includes invasive *N. neesiana* (Enderby & Koehler 2006). The best management approach for a given *S. plana* conservation site will always be one that is specifically tailored to the area and draws upon on-ground research and specific practical experience (New 2019). The management selected for a given site should be informed by a detailed understanding of its vegetative species composition and density, which can be achieved through ongoing monitoring (Kutt *et al.*, 2015) (Appendix C). Care should be taken to monitor the composition of the highest-quality *S. plana* habitat within a site, such as north-facing slopes dominated by *Rytidosperma* and *Austrostipa*, as these areas are likely to retain the core population reservoir of a given *S. plana* population, particularly during wetter years (Kutt *et al.*, 2016). Monitoring and managing *S. plana* populations must account for landscape heterogeneity, or risk missing subtleties in a population's distribution that are pertinent to its ongoing survival (Kutt *et al.*, 2016). *S. plana* management must be site specific and adaptive, rather than generalised and prescriptive, especially given the knowledge gaps surrounding its lifecycle and the limitations of current survey techniques (Kutt *et al.*, 2016.)

(Appendix D). Similarly, *S. plana* habitat will often extend beyond the boundaries of a specific management area (DEWHA 2009). These peripheral zones are relevant when considering the functionality of a *S. plana* population and its ecological needs. Management of these peripheral zones will often require broader stakeholder engagement and the development of an integrated approach across a wider area (DEWHA 2009; ACT Government 2020). The core management actions relevant to *S. plana* conservation include biomass control measures, weed removal and the reestablishment of native grasses (Mata *et al.*, 2017). To deploy these methods effectively, managers must have the freedom to adapt their approaches in a timely manner, responding to emerging local conditions (ACT Government 2020). Overly restrictive, generalised, catch-all approaches to *S. plana* site management, as with any species management, can prove detrimental to conservation outcomes (A. Kutt, E. Stone, N. Oddie, R. Bellchambers 2022, *pers coms.*) and can conflict with the social and economic practicalities underlying farm activities (Teague *et al.*, 2013).

Site management for *S. plana* should strive to retain a native grass sward in which the height of most of the leaf mass (except for seed-bearing tillers) is between five and fifteen centimetres tall, and comprises a high proportion of *Rytidosperma* and *Austrostipa* tussocks (ACT Government 2017c). This structure should retain an intermediate density and be interspersed with areas of bare ground (ACT Government 2017c). A degree of landscape level biomass heterogeneity is preferable, both in order to support other biodiversity assets at a site, such as *Delmar impar* (Howland *et al.*, 2004; ACT Government 2017c) and to retain the older, larger-sized individual c3 grass tussocks that *S. plana* favours (Gibbons & Reid 2013). Before assigning a biomass regime to a high biodiversity site, it is essential to understand the existing ecological processes in action (Lunt *et al.*, 2007). Hasty implementation of a novel broad-scale approach can upset the existing natural processes regulating the balances within a grassland site (Lunt *et al.*, 2007). Retention of such processes may be critical to its threatened taxa (Lunt *et al.*, 2007). In species-rich native grasslands, intervention through ecological management must retain and bolster these processes, rather than rapidly alter or replace them. If existing ecological processes are changed, it is likely that biodiversity declines will ensue as a result (Lunt *et al.*, 2007).

Native grazing and naturally occurring burns are optimal for biomass control in larger sites that retain these processes at the landscape scale (Mata *et al.*, 2017). The optimal density of kangaroos, to provide adequate biomass reduction without impacting biodiversity, is still a developing science (Antos & Williams 2015), and the appropriate intensity of natural grazing at a given site will vary depending upon management objectives. When kangaroos increase above about 2/Ha (Barton *et al.*, 2011), the intensified grazing will produce a low, even-structured grass sward (Antos & Williams 2015). However, this high intensity grazing negatively impacts bird, reptile and invertebrate abundance in grasslands (Antos & Williams 2015), making it suboptimal for areas with broader conservation goals (Lindenmayer *et al.*, 2016). In extreme cases, such as during drought conditions, kangaroo culls can be necessary in order to prevent over-grazing, and damage to the site’s vegetation structure (ACT Government 2017c). High biodiversity sites that have been frequently burnt over several decades will have an associated suite of floral species adapted to regular fires.

Such areas could incur biodiversity declines if the frequency or temperature level of their burning regimes is reduced (Bradstock & Auld 1995; Morgan 1998). Management of *S. plana* sites with high biodiversity values should seek to retain a close approximation of the site's historic fire cycle. While the retention of natural ecological processes is ideal, this is not always possible in fragmented landscapes, and more direct intervention through practical management becomes necessary. Moreover, many of the landscapes where *S. plana* has survived in high numbers in Victoria are the products of anthropogenic disturbance regimes, such as stock grazing (Sinclair *et al.*, 2014), and effective management must therefore maintain this modified *status quo* (Morgan 2015).

Maintaining a c3-dominated grassland at a low, uniform height, by mowing or slashing, produces excellent results for *S. plana* (e.g., Salisbury Bushland Reserve, Douglas 2004; Parsons Brinckerhoff Australia 2008). Mowing is particularly useful at dry, westerly sites where *S. plana* completes its breeding cycle before the onset of summer. At these locations, site managers can remove dead, native biomass in the early summer before it becomes a fire hazard, without impacting either the moths or seeding of their food plants (Douglas 2004). Since Salisbury Bushland Reserve also supports the February-emerging *S. selene*, the ideal time for mowing this site is in mid-December (Douglas 2004). Regardless of when the site is mown, mulched grass cuttings should be removed before the flight season begins, in order not to block inter-tussock spaces with biomass during *S. plana* reproduction (Parsons Brinckerhoff Australia 2008).

The removal of weed competition, and the resulting increase in *Rytidosperma* abundance, measurably increases *S. plana* population numbers (O'Dwyer & Attiwill 2006). Any long-term plan to manage a site for *S. plana* conservation should involve measures to detect new weed influxes, manage existing weeds and record overall changes in weed abundance (Biosis 2019b). *Phalaris aquatica* is a particularly problematic weed for *S. plana* and must be controlled at *S. plana* conservation sites because of its ability to swamp native food grasses and then remove the required inter-tussock spacing of a site (Gilmore *et al.*, 2008; Richter *et al.*, 2013). On large sites that retain high numbers of *Rytidosperma* and *Austrostipa*, there is little direct risk to *S. plana* from targeting and removing *N. neesiana*, which is a Weed of National Significance that presents a general threat to native grassland biodiversity (Urlus 2021). Pure stands of weeds can be targeted with broad-spectrum herbicides such as Glyphosate, and broadleaf weeds in higher quality native grass habitat, can be targeted with a selective herbicide such as Clopyralid (Douglas 2004). Annual grasses, in addition to *Lollium perene*, can be targeted with a foliar herbicide, such as Basta or Gramoxone (Abzeco 2017). Given that the impacts of herbicides on *S. plana* larvae are currently unstudied, the possible risks of herbicide use may be deemed unacceptable at particularly high value *S. plana* populations confined to fragmented sites (Douglas 2004). Similarly, Glyphosate is not suitable for use around high quality areas on *S. plana* sites, because *Rytidosperma* species are especially susceptible to this chemical, and they are readily poisoned by minimal exposure to spray drift (DAWE 2021a). Slashing and biomass removal are excellent means of controlling exotic grasses, if appropriately timed to prevent these species from setting seed (Backstrom & Forbes 2019). At more degraded sites, controlled stock grazing and ecological burning provide useful tools to control pasture grasses and to restore an optimal, low biomass site structure (O'Dwyer & Attiwill 2000).

Biomass control through ecological burning is data deficient for c3-dominated grasslands (Morgan & Salmon 2019) and the impacts of controlled burning on *S. plana* are largely speculative (Edwards 1994, *cited in* ACT Government 2017a; ACT Government 2017c). Well-timed burns allow the destruction of annual grasses before they set seed and can potentially destroy the seed residue that weed species deposit in the surface soil, aiding the restoration of such areas (Douglas 2004). In order to retain the heterogeneity needed to support broader biodiversity and to reduce potential risks to *S. plana*, it is preferable to burn grasslands patch by patch, over successive years, in a mosaic, with no single patch being burned more than once in three years (Urlus 2021). The preferred timing of burns in *S. plana* habitat is during March and April, during cool conditions (ACT Government 2017c; DAWE 2021a). A prudent approach, in the absence of robust published data on the direct impacts of fire to *S. plana* populations, is to avoid burning sites within central and south Victoria, during the months of September to January, in order to avoid exposing eggs, pupae and emerging adults to fire (DAWE 2021a). In the Wimmera, *S. plana* generally completes its flight season before December, potentially allowing sparing use of patch burns in early summer to eliminate weed seeds, at no risk to *S. plana* (Douglas 2004). If ecological burning is deployed to remove biomass at a larger scale, the programme should segregate the most optimised and essential *S. plana* habitat from the broader burn treatment. This segmentation should include any north-facing slopes dominated by *Rytiosperma* and *Austrostipa* (Kutt *et al.*, 2016). Such areas require greater care than the rest of the site, and ideally, they should receive targeted, location-specific small scale burning in order to reduce biomass, without threatening *S. plana*, or risking the loss of its food plants (Kutt *et al.*, 2016). Grassland burns should always be accompanied by follow-up weed control (Backstrom & Forbes 2019). The need for this kind of finer scale management, and its situational relevance, will vary from site to site, depending upon the wider management goals for the area (E. Stone, 2022, *pers coms.*). A greater level of targeted finesse is advisable, for example, at sites that support high priority *S. plana* populations, at isolated locations, or on the geographic edges of the species' range, where unique genetic haplotypes could remain (see Clarke & Whyte 2003). Notably, ecological burns are less effective at suppressing weeds in highly disturbed agricultural remnants with protracted grazing histories (Backstrom & Forbes 2019) and have a higher potential for off-target mortality of desirable grass species (Sinclair *et al.*, 2014). Burns in such areas produce disturbance and promote new influxes of weeds (e.g., Gorse and, *N. neesiana*), as a result, agricultural grazing land that doubles as *S. plana* habitat requires a higher level of post-burn weed control (ACT Government 2017c). Additionally, rabbit grazing can interfere with the recovery of native plants in post-burn areas, so rabbit control measures can sometimes become necessary as a result of burning (DAWE 2021a). In summary, ecological burning allows a timely and well-targeted approach to biomass reduction at *S. plana* sites, provided it is completed in a well-timed manner, appropriate for the site.

Pulse sheep grazing is an effective strategy for controlling c3 weedy grasses during the winter (Douglas 2004), with the stock subsequently removed at the end of winter, in order to allow native c3 grasses to set seed (DSE 2004). In general, c3-dominance in native grasslands arises as a result of past disturbance through intensive grazing (Morgan *et al.*, 2017; Morgan & Salmon 2019; J. Morgan, 2022, *pers* coms), which specifically favours the native c3 grasses that *S. plana* needs as food plants (Garden *et al.*, 2001; Dorrough *et al.*, 2004; Gibbons & Reid 2013). A study in New South Wales found that three *Rytidosperma* species associated with *S. plana* (*R. carphoides*, *R. erianthum* & *R. auriculatum*), declined in dominance when sheep grazing was replaced with macropod grazing (McIntyre *et al.*, 2022). Accordingly, many areas of past grazing land retain populations of *S. plana*, in the absence of any targeted management approaches (e.g., Dear 1997; Bainbridge & North 2007; Abzeco 2018). However, pulse grazing can fail to produce *Rytidosperma* dominance, if stock are left in place for overly protracted periods, because *Rytidosperma* species produce most of their new foliage during the early spring (Douglas 2004), and these green shoots are highly palatable to sheep (N. Oddie 2022, *pers coms.*). The impacts of grazing on *Rytidosperma* are harder to predict when soils are high in phosphorous (<12 mg/kg) (Mitchelll *et al.*, 2019).

Drier sites (such as those in the Wimmera) are likely to respond less favourably to grazing management and will require extended resting periods to allow desirable grasses to recruit (Teague *et al.*, 2013). Under optimal conditions, sheep should be removed from a *S. plana* offset before the end of winter, in order to maximise the chances of the desired outcome (Douglas 2004). In instances where sheep grazing causes a decline in native plants, this should trigger a resulting change to the duration and intensity that the blocks are grazed (Teague *et al.*, 2013; Wong & Dorrough 2015) or supplementation with mowing (Douglas 2004). Increasing the intensity of grazing by reducing paddock sizes and shortening grazing intervals allows more precise targeting of weed patches, while providing more time for desirable grasses to recover through germination and regrowth (Teague *et al.*, 2013). Grazing also provides a useful control technique to maintain *Phalaris aquatica* at a low density across a site (Biosis 2018a). However, sheep will avoid tall patches of high biomass *Phalaris*, which are better controlled through burning (J. Taylor, private grazier, 2022, *pers coms.*). More focused and intensive approaches are also required, if grazing is used to control other perennial grasses (e.g., *Holcus lanatus*), which sheep will normally avoid in favour of more palatable grass species (R. Bellchambers 2022*, pers coms.*).

If grazing is used to target *N. neesiana* then cattle, despite their higher capacity for soil disturbance (Greenwood & McKenzie 2001), are preferable to sheep (Mata *et al.*, 2017). Cattle are less likely to carry *N. neesiana* seeds on their bodies, which will reduce seed dispersal around the site (ACT Government 2017c). Cattle grazing can also provide an alternative approach to remove both dead, accumulated biomass and weeds that are unpalatable to sheep (Biosis 2019b). All biomass control using grazing should be carefully monitored to ensure that stock do not target palatable native grasses and forbs in a destructive manner (Douglas 2004). Care should also be taken whenever stock have access to topographic areas with an essential role as *S. plana* refugia, during wetter seasons (Kutt *et al.* 2016; T. Wills 2022, *pers coms.*). Grazing is more likely to cause long-term soil compaction, and its compounding impacts (e.g., soil desiccation, reduced root growth in food plants (Douglas 2004; ACT Government 2017c)), if it is deployed at particularly dry sites (Greenwood & McKenzie

2001), such as those in rain shadows or sites on the Wimmera Plain. Pulse grazing for biomass control can also promote the spread of broadleaf weeds (e.g., *Hypochaeris radicata* and *Arctotheca calendula*) which are not targeted by sheep and must then be manually controlled (DSE 2004). Grazing, in summary, has been a core process shaping the site structure of the areas used by most extant *S. plana* populations in Victoria and, if deployed in a site specific and adaptive manner, provides an excellent tool to maintain the habitat conditions needed by these populations. However additional approaches are required in particularly degraded habitats.

Sites smaller than 10 Ha in total size are proportionately less useful as *S. plana* reserves because they are too small to buffer a population from future modification events. At smaller sites, even relatively minor habitat changes could cause population extinction (DEWHA 2009). At smaller and more degraded sites, deliberate retention of *N. neesiana* patches is an appropriate strategy to protect particularly significant *S. plana* populations (Mata *et al.*, 2017; Backstrom & Forbes 2019). Mata *et al.* (2017) identified predation risk as the single biggest factor influencing adult *S. plana* survival in urban reserves and suggested that its impacts on fragmented populations may be underappreciated and in need of more active management. Accordingly, small urban reserves designed purely for *S. plana* conservation should minimise the use of fences and other structures, in order to avoid providing perching or nesting habitat for avian predators of adult *S. plana* (DEWHA 2009). Fences should be designed to allow easy passage for adult *S. plana*, while keeping available perching surfaces to a minimum (DEWHA 2009). Sites in urban areas can also accumulate rubbish, including yellow glass objects that interfere with *S. plana* reproduction, so these areas require routine clean-ups, particularly prior to slashing (Bainbridge & Longmore 2016). Additionally, rabbits should be controlled within *S. plana* reserves, in order to prevent this pest species from increasing to high densities, causing erosion and declines in vegetative structure (DAWE 2021a).

Whenever degraded rural areas are selected for restoration as *S. plana* habitat, care should be taken to choose sites with appropriate topographies, landscape positions and historic vegetative communities, in order to best support *S. plana* (Kutt *et al.*, 2016). Reclaimed agricultural areas that are intended as *S. plana* habitat may be unsuitable due to high nitrate and phosphorus levels in the soil resulting from soil enrichment (O' Dwyer & Attiwill 1999). Such areas may occur as peripheral habitat adjoining unimproved grassland with resident *S. plana* (e.g., Craigieburn Grassland, DNRE 1998; Bainbridge & North 2007). Phosphorous specifically should not be over 14 mg per kilogram (O' Dwyer & Attiwill 1999). However, some of the *Rytidosperma* species that are most associated with *S. plana* (*R. carphoide*s (Edwards 1993, *cited in* Richter *et al.*, 2013b), *R. auriculatum* (Rowell 2013), *R. erianthum* (O' Dwyer & Attiwill 2000)) are fertility-intolerant and unlikely to thrive when soil phosphorous levels exceed 6 mg per kilogram (Mitchell *et al.*, 2019). Generally, soil scalping is required if such sites are to be remediated at relevant timescales, which requires the use of heavy machinery (Gibson-Roy & Delpratt 2015). Ecological burns can assist in reducing the phosphorous and nitrate loading of the soil at a site (Biosis 2019b) and nutrient stripping by removal of weedy biomass from target areas can assist with nitrate reduction (Gibson-Roy & Delpratt 2015). Nutrient stripping can either be achieved through manual weed removal (Gibson-Roy & Delpratt 2015) or by careful management of stock grazing (N. Oddie 2022, *pers coms*.). After restoration is complete, *S. plana* should then be able to recolonise such areas during optimal breeding seasons if they have access to connecting habitat corridors (Brown *et al*., 2012; New 2012; Kutt *et al*., 2015; Bainbridge & Longmore 2015).

Chapter 7. Case studies

This chapter describes an example site, Chepstowe, where a population of *S. plana* is managed on a farming property. Chepstowe is a historic family sheep farm in the Victorian Midlands, with an emphasis on biodiversity preservation through low intensity land use (Williams *et al.*, 2015). Several other private graziers were also interviewed, regarding their experiences managing S. plana habitat in an agricultural context. All these farms contained blocks of delineated S. plana habitat, that were established as offsets for *S. plana* populations removed through urban and agricultural expansion within and near to Melbourne (see subchapter 5.2). Additionally, six Parks Victoria and Hume City Council reserves that support *S. plana* are discussed in Appendix E.

7.1 Case study 1: Chepstowe

Chepstowe is a textbook example (Williams *et al.*, 2015) of agricultural management that is conducive to maintaining *S. plana.* The 500 Ha Oddie family farm at Chepstowe in the Victorian Midlands was established in the 1860’s with a philosophy of “low input, low impact” land use (Williams *et al.*, 2015; Biosis 2018b). The property lies approximately 150 km to the west of Melbourne and comprises a landscape of gently rolling hill country that supports heavily modified Victorian Volcanic Plains (VVP) grassland and grassy woodland habitat (Abzeco 2018). The current owner, Neville J. Oddie, assumed management in 1979, and has sought to retain, through voluntary conservation covenants, biodiversity-rich remnants on previously uncropped and unfertilised areas of this property (Williams *et al.*, 2015; Biosis 2018b). After these remnants were fenced off, a native grass sward dominated by *Rytidosperma* spp., *Austrostipa* spp. and *Themeda triandra*, gradually regenerated over the next 10–15 years (Williams *et al.*, 2015). In addition to *S. plana*, these areas support various other threatened taxa including Clover Glycine (*G. latrobeana*), Fat-tailed Dunnart (*Sminthopsis crassicaudata*), Striped Legless Lizard (*D. impar*) and the Hoary Sunray (*Leucochrysum albicans* subsp. *tricolor*) (Abzeco 2018).

Oddie’s ongoing landscape management has endeavoured to harness and apply the natural preferences of his sheep flocks around the year rather than restricting their movements with fences (Williams *et al.*, 2015), in the belief that sheep grazing can support environmental outcomes while simultaneously fulfilling production objectives (N. Oddie 2022, pers coms.). Specifically, Oddie is aware that his flocks prefer to make encampments on high ground, in shaded areas, and that they travel to dams in low-lying areas of the farm in order to drink (N. Oddie 2022, pers coms.). Essentially, Oddie’s sheep flocks reduce the nutrient loading of high biodiversity zones on the property, through nutrient stripping, first by removing palatable weeds, and then by depositing their nutrient load at degraded camp sites elsewhere on the farm (N. Oddie 2022, pers coms.). Oddie’s approach contrasts a popular grazing regime trend amongst sheep graziers that emphasises intensive crash grazing of smaller areas, over periods spanning a few weeks (N. Oddie, 2022, pers coms.).

Oddie’s farm has several ten-year Offset Management Plans (OMP), in place for his various remnants, which include three areas of Temperate Grassland of the VVP, which are managed and monitored by external contractors (Williams *et al.*, 2015). Each OMP includes a quadrat-based vegetation survey at the beginning, middle and end of the assigned works, as well as targeted flora surveys after burns (Williams *et al.*, 2015). Oddie (2022, pers coms.) states that he requires that any new OMPs proposed for his farm do not disrupt his general operations, permit stocking on the offset area and produce measurable environmental benefits. Oddie's farm is biodiversity rich. When compared to the surrounding agricultural matrix, which consists of broadacre crops and introduced pastures that offer little connectivity for grassy ecosystem biota, it is clear that the value he places in biodiversity function results in a very different landscape to more traditional agricultural practices that exclusively prioritise production outputs (N. Oddie 2022, pers coms.). Nevertheless, Oddie (2022, pers coms.) sees a growing interest in biodiversity amongst Victoria's farming community, who recognise the opportunity to learn something new about their properties, in addition to seeing the potential for economic compensation from ecological offsetting.

S. plana is one of many ecological values present at Oddie's farm, and he is less interested in the specific numbers of this *S. plana* population than in the overall health of the broader landscape, and how he can best manage his land to enhance this (Williams *et al.*, 2015). Oddie (2022, *pers coms.*) first became aware of *S. plana* on his property around 2010 (ABZECO 2018). The property received targeted surveys for *S. plana* in 2015, by an external contractor, which confirmed the abundance of *S. plana* across his property and affirmed the farm's general suitability as a site for *S. plana* offsetting (Abzeco 2018). This survey work also produced a distribution map of the species' site occupancy (Abzeco 2018). *S. plana* currently occurs in three areas of the property, each of which are linked to various offsets, although the species is also patchily distributed across the broader property (Abzeco 2018, *Figure* 9), with the notable exception of the sheep camps (N. Oddie 2022, *pers coms.*). The private properties that surround Oddie's farm have been inadequately surveyed for *S. plana*, and it is likely that additional local populations remain undiscovered (N. Oddie 2022, *pers coms.*). Accordingly, past assessment of his property has recommended extending *S. plana* surveys to nearby roadsides and adjacent areas of habitat (Abzeco 2017).

Figure 9. offset habitat at Chepstowe, predominantly maintained through low-density sheep grazing. May 2022. (Photo credit: D. Harmshaw 2022).

The ten-year OMPS in place on Oddie's farm require him to improve the habitat quality of *S. plana* offsets by increasing native grass cover and reducing weeds through stock management, weed control, biomass management, rabbit control and management of tree and shrub recruitment (Biosis 2019b). Throughout the managed offset period, the sites receive biennial *S. plana* surveying and vegetation assessment for weed and native grass cover, which form part of Oddie's annual reporting requirements (Biosis 2018b; Biosis 2019b). Progress at the offsets is reviewed by Trust for Nature, who undertake occasional site audits of each offset zone, throughout the 10-year period (Biosis 2019b). After the completion of each OMP, these sites will then transition to permanent Trust for Nature Conservation Covenants (Abzeco 2018).

The farm's biomass changes markedly throughout the year, and the microhabitats of different zones across the property all have different levels of biomass (N. Oddie 2022, *pers coms.*). Some of the highest *S. plana* densities at Chepstowe occur in areas of the farm that have never received active management for this species and have been grazed year-round, for several decades (*Figure* 10. N. Oddie 2022, *pers coms*). Past survey work has established a density of 29.6 *S. plana* per Ha at one of his oldest *S. plana* offset zones (Biosis 2018b), with the species readily occupying weedy habitats across the property (N. Oddie 2022, *pers coms.*).

Figure 10. *S. plana* occupies this grassy hillside at high density, despite decades of year-round sheep grazing and no specific management actions to maintain this species. May 2022 (Photo credit: D. Harmshaw, 2022).

Oddie favours grazing to manage biomass on his *S. plana* offsets, as he considers this approach to have a lower ecological impact than burning (Williams *et al.*, 2015) as well as being a relatively simple technique for him to target towards different areas of his farm around the year (N. Oddie 2022, *pers coms.*). Because the site has heavy clay-rich soil, Oddie can deploy relatively high stocking rates compared to what would be possible on more erosion prone soil types elsewhere in Victoria (N. Oddie 2022, *pers coms.*). Oddie (2022, *pers coms.*) favours gentle, natural grazing from late summer to early spring, and sees little need for more intensive high-density crash grazing, which has the potential to impact the broader biodiversity values of his property. Oddie (2022, *pers coms.*) does not rotate his flocks according to any specific schedule, and stock availability to reduce grassy biomass is determined by the broader farm work that he performs throughout the year. His flocks are removed from the *S. plana* offsets during the months of September–February (N. Oddie 2022, *pers coms.*). Although his OMPs afford Oddie a high degree of flexibility in terms of grazing, he is required to maintain detailed records of the stock numbers and durations involved (Biosis 2019b). Oddie (2022, *pers coms.*) envisages drastically reducing his personal flock size in later years as he pursues other endeavours, though notes that even during years where he has no stock present, he will maintain the grazing regime on the property with stock brought in from other farms.

Ecological burning removes dense patches of dead biomass that would not otherwise be grazed by sheep (N. Oddie 2022, *pers coms.*), depletes the seed banks of weed species, reduces soil nitrate and provides a back-up measure if grazing fails to achieve the desired outcomes for that year (Biosis 2019b). Deploying cool burns across the farmland interrupts woody plant encroachment into the grasslands, such as leguminous native *Acacia* spp. and exotic gorse (*Ulex europaeus*) (N. Oddie 2022, *pers coms.*). Oddie' s *S. plana* OMPs restrict him from burning from November–January, during the *S. plana* flight season, and each offset can only be burnt once in a three-year period, without external approval (Biosis 2019b). Oddie (2022, *pers coms.*) considers an overly prescriptive approach to ecological burning to be impractical, as well as potentially counterproductive to ecological outcomes, if local conditions have hampered other biomass reduction measures throughout the year. He emphasises the need for burn timing to be drawn from real-time, site-based information, rather than external mandates (Oddie 2022, *pers coms.*). He looks to local weather conditions, as well as the week-to-week practicalities of his farm work, to dictate when he burns an area (Oddie 2022, *pers coms.*). In contrast with his purely *S. plana*-targeted offsets, ecological burning provides the primary method of biomass reduction used on the highest quality remnant patches of VVP grassland on his property (Williams *et al.*, 2015). Oddie (2022, *pers coms.*) prefers to avoid combining burning and grazing and is not permitted to graze his *S. plana* offsets until six months after an ecological burn, without external permission (Biosis 2019b).

These broad management approaches favour specific weed species, which then require targeted management. Sheep camps cause broadleaf weeds such as Spear Thistle (Cirsium vulgare) to proliferate, which require local weed control with selective herbicides such as Clopyralid (Oddie 2022, pers coms.). Due to their unpalatability to sheep, two years of above average rainfall driven by La Niña, combined with grazing and burning exclusions during spring months has enabled Brown Top Bent Grass (*Agrostis capillaris*) and Creeping Bent Grass (*A. stolonifera*) to develop into major problems in some areas of the property, (Biosis 2018b; N. Oddie 2022, *pers coms.*). These perennial C3 grass species have densely swamped remnant patches of *Rytidosperma* utilised by *S. plana* (N. Oddie 2022, *pers coms.*). Oddie' s (2022, *pers coms.*) current strategy is to contain the infestations from further spread. His most recent OMPs specifically require that these species are controlled as high priority weeds, variously suggesting crash grazing (Biosis 2019b), patch burning and follow-up herbicide application as management techniques (Biosis 2018b). Grazing has proven inadequate for *Agrostis* management however, and Oddie (2022, *pers coms.*) currently sees potential for incorporating slashing as an alternative weed control. He also envisages implementing a formal monitoring project to assess the impacts of *Agrostis* in more detail and to explore potential control measures across his property (N. Oddie 2022, *pers coms).*

Small pockets of the farm are permitted to return to shrubland dominated by Blackwood (*Acacia melanoxylon*) and Hedge Wattle (*A. paradoxa*), which Oddie considers to be the natural state of these landscapes (N. Oddie, 2022*, pers coms.*). He remains curious about the potential of these zones to transition towards River Red Gum (*Eucalyptus camaldulensis*) woodlands and the optimal natural balance for areas of woodland for the farm. Consequently, he has fenced off a regenerating woodland area adjacent to his newest *S. plana* offset, seeking to provide habitat for other native species (N. Oddie 2022, *pers coms*). The *S. plana* offsets themselves must be maintained as open sward grasslands in order to maintain *S. plana* habitat, although one tree or shrub per 0.5 Ha may be retained in order to provide habitat for native fauna (Biosis 2019b).

Oddie is concerned by a lack of integration across agricultural sciences, practical biodiversity management, and the regulatory structures that govern these fields, which conflicts with his own holistic approach to farming (Williams *et al.*, 2015). While Oddie (2022, *pers coms.*) encourages landowner participation in conservation, he nevertheless sees a pervading need for more direct application of conservation policy in order to protect the biodiversity of Victoria's agricultural landscapes. He notes that a landowner's initial participation within most conservation endeavours is currently voluntary, rather than forming a mandatory baseline tenet for general land ownership and agriculture (N. Oddie 2022, *pers coms.*). Accordingly, Oddie (2022, *pers coms.*) sees a need for broader government directives on biodiversity management within agriculture and an increased role for private consulting agencies to contribute within this space, both as advisors for landholders and as auditors for local bodies. Oddie's approach to farming is a proven test case for *S. plana* management in rural Victoria, demonstrating the resilience of *S. plana* to low intensity farming practices and the need for flexibility and adaptability while managing *S. plana* habitat.

7.2 Additional case studies: Agricultural properties with *Synemon plana* offsets

In addition to visiting Neville Oddie's farm at Chepstowe, TREC Land Services completed five interviews with agricultural landowners in the Victorian Midlands, who were invited to share their thoughts and experiences on farming in association with *S. plana*. Additionally, two further agricultural landowners with *S. plana* on their properties, in the Central Uplands, contributed written responses to a short list of questions via email. All of these properties contained specifically assigned offsets for the protection of *S. plana*, established and funded to compensate for the removal of *S. plana* habitat elsewhere within Victoria. These sites were selected because they retained *S. plana*, and flora surveys had established suitable conditions to maintain these populations in perpetuity through ongoing management. The specific management activities and reporting obligations required for each offset were individually defined through separate OMPs. Typically, these included established targets to improve the cover of native grasses within the offset, and in most instances, also measured success through an increase in *S. plana* numbers. TREC Land Services also incorporated perspectives from attendees of Trust for Nature's conference event "Native Grassland Management on Farms in south-western Victoria" on the 16th of July 2022. All of the offsets discussed are covered by Trust for Nature Covenants. Most respondents preferred to comment anonymously. Accordingly, their contributions have been collated and summarised below.

Without exception, all areas selected for *S. plana* offsets were habitat associated with sheep farming that had experienced a history of light sheep grazing since European colonisation. These sites had never been treated with superphosphate or been subject to pasture enrichment. Generally, each property's specific *S. plana* offset sites had been managed in a similar way to the adjacent farmland on each property. In two such instances, *S. plana* were abundant across the entire properties, rather than concentrated within the specifically defined (and specifically managed) offsets, implying the low-density sheep grazing across these farms to be generally amenable to the survival of *S. plana*.

The other respondents seldom observed *S. plana* either within or outside of their assigned *S. plana* offset blocks but managed their broader properties in a similar manner. This meant that the suitability of the specific offset sites on these farms, as *S. plana* habitat, tended to reflect generally amenable land management approaches, rather than the `localised conditions (e.g., floral diversity, topography) within the specific blocks that had been selected for offsetting.

Most land managers expressed frustration at specific requirements within their OMPs that they did not feel were realistic annually deliverable expectations. Several respondents highlighted the difficulty of identifying unrealistic management requirements incorporated within the text of an OMP. Several OMPs lacked practical application advice to produce the required outcomes. The landowners had accordingly needed to engage third party conservation managers to assist them in delivering the agreed-upon outcomes. There was a prevalent concern that the OMP drafting process was too removed from the practical realities of land management and a perception that OMP drafting represented a legal framework to restrict certain activities on the offset, rather than providing any relevant advice to improve the offset’s conservation values. One respondent considered that the OMP drafting process was limited by the author’s requirement to comply with both State and Commonwealth auditing requirements, which further removed them from the practicalities of land management. The paucity of conservation management advice on achieving the outcomes specified by the OMPs left some landowners struggling to deliver on annual outcomes. In two instances, problematic weeds (*Agrostis* spp.) had been missed by the consultant who had initially assessed the property. The resulting OMPs did not factor the impacts of these weed species, or the resources required to control them, into their mandatory outcomes. The landowners had then needed to consult third party conservation managers for advice on weed control. Some interview subjects felt that they had been given insufficient input into the OMPs assigned for their properties, or that their OMP was too generic in nature to address specific issues inherent to their properties. These included safety issues, vehicle access and production requirements. One respondent considered that their OMP’s requirements for stocking numbers did not adequately account for varying moisture and biomass on the site. Three respondents asserted that their OMP requirements were so restrictive that they had caused a resulting decline in the habitat condition of their offsets compared to their prior use, purely as grazing land. This was generally due to emergent weeds and changed biomass levels.

In two instances, landowners felt that ten (and in one example, twenty) years was a too long timespan for a legally binding commitment of this kind, given the lack of site specificity present within the OMPs. These respondents asserted that neither signatory party could predict local site factors with sufficient reliability to be able to commit to such an outcome with confidence. The respondents cited a failure in their OMPs to account for mid-term changes to weather and annual temperature. Several respondents highlighted concerns with OMPs that directly measured success through higher numbers of *S. plana* counted during surveys, which they felt was outside of their control. The monitoring requirements for each offset varied considerably, with one landowner stating that they had no mandatory requirement to monitor the *S. plana* population on their block, and others legally required to submit biennial survey reports.

Ecological burns were commonly deployed to target weedy grasses that were unpalatable to sheep (such as *Agrostis* spp.), and generally achieved good results if they were timed well. One respondent noted that, after hot, dry summers, even autumn burns often needed to be delayed until May due to safety concerns and another respondent considered a three-year burn cycle to be ideal for biomass management on a grazed site. Others preferred avoiding burns due to concerns that they would promote weed recruitment or cause negative impacts to biodiversity. Restrictions to spring and early summer burning were a common concern, with several respondents expressing scepticism on the benefits that this brought to *S. plana*, when compared with the limitations it imposed on weed control within the offsets. The timing and communication requirements specified in some OMPs rendered ecological burning effectively impossible at some offsets, despite burns being listed as a core biomass reduction and weed control approach within the plan. The lack of site specificity in many *S. plana* OMPs meant that vehicle access to specific areas or site-specific safety concerns, were seldom incorporated. Respondents recognised the need for rapid re-treatment of post-burn zones for emerging weeds, which disinclined one respondent towards prescribed burning.

Several respondents considered light, continuous sheep grazing as a particularly effective strategy for controlling *Phalaris aquatica*. Crash grazing was deployed to target *Holcus* and to reduce overall biomass levels. One respondent highlighted a stipulation in their OMP that they needed to apply to DELWP in writing before crash grazing their offsets, despite crash grazing being listed as an agreed-upon management approach within the plan. The respondent considered this condition to be inconsistent with adaptive site management, under changing conditions. The use of either old ewes or wethers was favoured to graze an offset, rather than productive stock, in order to reduce impacts upon the farms' production outcomes. While conditions remained too dry to burn safely, summer grazing, at the end of the *S. plana* flight period, was seen as a necessary fuel reduction measure. One respondent stated that an exclusion from grazing their *S. plana* offset, during November, had proven problematic, because they considered this to be the key time of year for removal of accumulated grass biomass. While no cattle graziers were interviewed, most respondents expressed scepticism regarding the use of cattle for selective weed control within a VVP grassland context.

Interview respondents felt that they would have been better able to produce conservation outcomes for *S. plana* if they had personally been included in the offset draft process. Several respondents expressed reservations regarding the future management of their *S. plana* offsets after the completion of the OMP. Wherever OMPs restricted landowners from making management decisions, based on local weather and site conditions, the respondents felt unable to manage their offsets to a suitable standard. The interview subjects emphasised the need for *S. plana* offsetting to allow for variable and flexible approaches to land management, in order to allow site managers to respond to local changes in an adaptive manner. It was noted that earlier *S. plana* OMPs, which had focused on maintenance of ecological values, had produced more realistic conservation outcomes, than a current trend within OMPs, which typically measure success through a documented increase of the offset's net ecological assets. The best and most stable outcomes for *S. plana* conservation will always be achieved when management is tailored toward the conditions of specific sites, draws upon the knowledge and practical expertise of their managers, while allowing rapid and reflexive responses to changing conditions. The current science that underpins the management of *S. plana* in rural landscapes was, for the most part, developed elsewhere (Kutt *et al.*, 2015). Knowledge drawn from studies of fragmented urban *S. plana* populations will be less applicable to agricultural landscapes, which necessitates a less prescriptive approach to management (Kutt *et al.*, 2015).

Chapter 8. Future research directions

Future research on *S. plana* must seek to quantify the response of the species to ecological burning (DAWE 2021a) and to different timing, duration and intensities of grazing (Kutt *et al.*, 2015), particularly during the flight season. A lack of knowledge on *S. plana*'s capacity to diapause within the soil between years (Richter *et al.*, 2013b), confounds attempts to map populations with reliability, particularly during wet summers (Kutt *et al.*, 2016). There remains a pressing need for formal identification keys for both *S. plana* pupae and the different instar stages of the species (Richter *et al.*, 2013b). An improved understanding of *S. plana*'s larval ecology will provide better options for reliable translocation, to bolster fragmented populations of high conservation value (ACT Government 2017c). The genetic diversity of *S. plana* is currently overdue a detailed reanalysis (Clarke & O'Dwyer 2000), due to the abundance of new populations that have been discovered over recent decades (DAWE 2021a). Such research may then facilitate new approaches to *S. plana* surveying, such as environmental soil analysis for *S. plana* DNA (N. Noble, EnviroDNA, 2020, pers coms.).

8.1 Developing areas for research

8.1.1 Research Priority #1. Pupal Case Identification and Larval Keys.

Improving knowledge of the early stages of the *S. plana* lifecycle will improve survey efficiency, allowing for more reliable field detection under time pressure. This will help to safeguard viable populations from development, while helping ensure that populations lost to development are correctly documented and appropriately offset elsewhere.

In 2014, the University of Canberra successfully maintained *S. plana* larvae in a glass house, for a period of nine months, achieving a 75% survival rate, before translocating the cohort to a new field location (Sea & Downey 2014, *cited in* ACT Government 2017c; Brawata *et al.*, 2017). Less the 5% of these individuals emerged as adult moths (Sea & Downey 2014, *cited in* ACT Government 2017c; Brawata *et al.*, 2017). Historically, the inability to study larvae *ex situ* (Douglas 2004), has perpetuated knowledge gaps surrounding the larval and pupal ecology of *S. plana*, and uncertainties regarding the lifecycle, longevity and synchronicity of the species (Richter *et al.*, 2013b). These uncertainties have bled into survey design and requirements, and influenced site management approaches (Kutt *et al.*, 2015). Specifically, surveys are unable to account for larval or pupal diapause (DEWHA 2009; DSE 2004) and may produce misleading results if larvae indeed sequester themselves into temporally isolated annual cohorts (Kutt *et al.*, 2015; Richter *et al.*, 2013a; Enderby & Koehler 2006). The early life stages of *S. plana* require formal description, so that suitable identification keys can be made available for land managers and consultants. Such a key would also need to describe other common soil larvae and pupae present in south-eastern Australian grasslands that are likely to be misidentified as *S. plana*. Pupal case collection is currently all but unheard of as a survey method within Victoria. The lack of a readily available pupal case key for *S. plana* surveys directly limits the quality of surveying that is practically possible, particularly during years of above average rainfall. If it is to be adopted as a more standard survey approach, pupal case identification would benefit from a standardised detection model, allowing surveyors to balance detection rates of pupal casing against the total emergence rate of adults (Richter *et al.*, 2013a). In a more general sense, Time to Detection Models also present an efficient method to streamline *S. plana* survey results during suboptimal conditions (Kutt *et al.*, 2015).

Pupal case collection holds additional potential for identifying the host plant that was originally used by emerged adults during their larval stages (Richter *et al.*, 2013a) (Appendix D3). Stable isotope analysis (Tibbets *et al.*, 2008), has previously allowed identification of the specific host plants utilised by *S. plana*, through tissue analysis of collected pupal casings (Richter *et al.*, 2013a).

8.2 Additional areas of research

8.2.1 An updated genetic library for *Synemon plana*

A detailed genetic library for known populations of *S. plana* will allow the identification of priority populations possessing unique genetics, permitting conservation resources to more effectively safeguard the resilience and diversity of the species into the future.

Creating a new genetic reference library for *S. plana* will directly benefit the species, as increased knowledge on the genetic diversity of Victoria’s *S. plana* population would allow for more targeted application of conservation funds towards high priority populations (Clarke & O’Dwyer 2000). These genetically distinct subpopulations are most likely to occur on the edges of *S. plana*’s distribution or at sites with notably different topography and ecological processes (DAWE 2021a). *S. plana* is currently considered to have arisen in central Australia, before gradually moving west across Victoria, and then northwards along the Great Dividing Range (Clarke & Dear 1998; Edwards *cited in* Clarke & Whyte 2003). Existing genetic information has so far highlighted the significance of *S. plana* populations in the Wimmera, implying these to be of particularly ancient origin (Clarke & O’Dwyer 2000; Clarke & Whyte 2003). Within a Victorian context, sites occurring within the Wimmera and the northern Riverina are a priority for further study, due to their isolation from other extant populations and locally distinct habitat conditions. Similarly, grassland habitats throughout eastern Victoria have lacked survey attention for *S. plana* and may retain genetically isolated populations of high conservation priority. *S. plana* populations are noted for being increasingly related with geographic proximity due to the limited dispersal ability of females (Clarke & O’Dwyer 2000; DEE 2016). Accordingly, populations to the north-east of Victoria have an increased chance of genetic relatedness to the more recently emerged and less genetically diverse populations of NSW and the ACT. However, any area where c3 grass dominance has arisen independently of disturbance processes (e.g., frost hollows, rain shadows) holds potential for comparatively ancient *S. plana* populations, allopatrically separated from more generalised populations nearby, and is worthy of attention.

Such a reference library has the additional benefit of forming a necessary first step within the development process required to produce a reference assay for environmental soil sampling (N. Nobel 2022, *pers coms.*). While applying environmental DNA sampling to soil plugs is a relatively new field within environmental sampling (N. Noble 2022, *pers coms.*), *S. plana* is likely to be a suitable species for such an approach, because it tends to favour specific topographical features within a site as refugia. Environmental DNA sampling would provide a useful supplement to presence/absence surveying, provided that it can be deployed in a cost-effective manner (N. Noble 2022, *pers coms.*). In contrast, deploying environmental DNA sampling at a sufficient scale to provide population data would be extremely expensive, reliant upon occupancy modelling and would potentially involve inappropriate levels of soil disturbance, particularly within small urban sites (N. Noble 2022, *pers coms.*). Assembling a suitable genetic sample for *S. plana* soil sampling would initially require the collection of one or two specimens from each major geographic region within the distribution of the species (N. Noble 2022, *pers coms.*). These samples could be complimented with non-invasive extractions from museum specimens and pupal casings collected within three months of adult emergence (N. Noble 2022, *pers coms.*). The resulting reference assay would then be robust enough to account for genetic variation within *S. plana*, while reliably distinguishing the species from sister species such as *S. selene* and *S. collecta* (N. Noble 2022, *pers coms*). Soil sampling would also allow for more general species lists to be assembled to catalogue the sub-surface biodiversity of each site surveyed (N. Noble 2022, *pers coms.*). EnviroDNA is currently developing genomic mapping techniques for a host of cryptic soil taxa, including arthropods, nematode worms and fungi (N. Noble 2022, *pers coms.*). The company sees potential in a more integrated approach to biodiversity surveying, involving the collection and assembly of site-specific biodiversity data derived from surveys targeted at single species, such as *S. plana* (N. Noble & S. Hale 2022, *pers coms.*).

8.2.2 Improved understanding of grazing and fire as management tools

Current approaches to S. plana site management are overly cautious due to a lack of robust quantitative data. Establishing firmer thresholds for S. plana’s resilience to different grazing and fire intensity, timing and frequency will allow more efficient use of resources and reduce conflicts between broader conservation goals and other land use priorities in managed S. plana habitat. This will further improve the appeal of S. plana conservation to stakeholders, creating more opportunities to preserve and bolster extant populations.

There is a pressing need for a more integrated understanding of the abiotic environmental factors that determine optimal conditions for *S. plana*, and how these change between flight seasons, in different regions of its range (Kutt *et al.*, 2015). These uncertainties reduce the efficacy of many active management programmes and, in extreme scenarios, can lead to ingrained management requirements that lack the flexibility needed to address and respond to emerging environmental changes (Kutt *et al.*, 2014; Mata *et al.*, 2017). Much of the current literature on *S. plana* management has been derived from urban sites, and accordingly, there is a prevalent need for more detailed research into the impacts of landscape scale heterogeneity upon *S. plana* population health and habitat occupancy, within agricultural landscapes (Kutt *et al.*, 2015). As management tools for *S. plana*, both ecological burning (DAWE 2021a) and livestock grazing (Kutt *et al.*, 2015) remain data deficient and quantitative studies exploring the potential impacts of both approaches to biomass management during the *S. plana* flight season should be a priority for future research (Kutt *et al.*, 2015; DAWE 2021a). DAWE (2021a) identify a need for quantitative data on how ecological burning impacts *S. plana* habitat occupancy, abundance, individual mortality, ranging behaviour and breeding success. Kutt *et al.* (2015) recommend targeted grazing studies incorporating experimentally controlled plots to produce quantitative data on optimal stock density and durations, under different environmental conditions. A wealth of suitable areas across Victoria are available for such research, with many participants in this review expressing a desire to contribute to future research initiatives and offering the use of the sites they manage for such endeavours. Similarly, there is a need for nationwide modelling of the influence of climate change on the distribution of *S. plana* (DAWE 2021a). This modelling will allow the identification of areas within Victoria where the species has high resilience. These areas of high resilience will likely act as long-term 'refuges' during changing climactic conditions and allow for strategic allocation of conservation resources across future decades (DAWE 2021a).

8.2.3 *Synemon plana* translocation.

Translocation provides a final option to bolster fragmented *S. plana* populations identified as high conservation priority, due to distinctive genetics.

Particularly isolated populations of high conservation priority may require additional bolstering through translocation of individuals to nearby suitable habitats (DAWE 2021a). It is critical that *S. plana* translocation programmes do not mix individuals from geographically isolated populations, in order to avoid outbreeding depression, or the loss of localised genetic haplotypes through non-random mating and natural selection (DAWE 2021a). *S. plana* translocation, via soil plugs, failed when attempted at Sheoak, Yea, due to wet weather, soil disturbance and a failure to re-establish native grasses (Jellie *et al.*, 2014). However, mixed success was achieved, during a *S. plana* translocation completed at Kinlyside Nature Reserve, Canberra (ACT Government 2017b; SMEC 2016 *cited in* Brawata *et al.*, 2017). In this project, larvae and pupae were hand collected and placed into drill holes within translocated habitat plugs (ACT Government 2017b). Although the long-term viability of this translocation has not yet been determined (ACT Government 2020b), high numbers of emerging adults were recorded from the translocated plugs in the following flight season (ACT Government 2017b), despite increases in *Phalaris aquatica* and *Nassella trichotoma* within the soil of the translocated plugs (Rowell 2019). The suitability of this approach is of course contextually dependent upon the specific weed species present at the donor and receptor sites, and the comparative importance of the *S. plana* population.

Conclusion

Since *S. plana* began, once again, to be recorded in grasslands across Victoria during the early 2000’s, the species has become a flagship for Victorian Volcanic Plains (VVP) grassland conservation (New 1997; New 2011; Antos & Williams 2015). *S. plana*’s status as a conservation flagship partly rests on the species being considered “largely […] restricted to grassland environments and dependent on them” (New 2019). However, recent academic findings and practical management experience have highlighted *S. plana*’s shortcomings as an indicator of intact, biodiversity-rich VVP grassland habitats. Far from being dependent upon such pristine habitat, *S. plana* thrives within moderately disturbed VVP grassland landscapes that retain a significant cover of native c3 grasses. For the most part, the native c3 grass swards remaining within lowland VVP grassy landscapes represent an intermediate stage of ecological succession, prior to the re-establishment and emerging dominance of *Themeda triandra*, a species that is more heavily impacted by sheep grazing. *S. plana* habitat in southern Victoria is now most often found within grazing land. In these modified grassland landscapes, the disturbance that is required to maintain the structure favoured by *S. plana* has arisen as a result of anthropogenic land use, where it is typically maintained at greater scales than occurs within more intact grasslands. Prior to European settlement, the structure and species composition of VVP grassland landscapes was maintained by aboriginal burning, which in turn replaced the grazing activity of extinct megafauna (Flannery 1994). In contrast, within the semi-arid regions of the state, the natural c3 dominated grasslands occupied by *S. plana* represent more stable deep-time systems produced and maintained by abiotic factors.

Historically, *S. plana* has been proposed as an “umbrella species”, whose conservation confers broader benefits to native invertebrates (New 1997). This assertion is difficult to defend given the habitat preferences of *S. plana* and the intensive biomass reduction measures needed to maintain its populations. Despite two decades of intensive survey work and conservation efforts targeted toward *S. plana*, scientific knowledge of the broader native invertebrate community associated with Australia’s temperate grassland landscapes remains at a rudimentary state (New *et al.*, 2011; Antos & Williams 2015; New 2019). Most invertebrate species native to Victoria’s grasslands remain formally undescribed, with their lifecycles and distributions all but unknown (New *et al.*, 2011; Antos & Williams 2015; New 2019). The creation of a system of targeted reserves for *S. plana* within Victoria, has not, for the most part, led to corresponding initiatives for Victoria’s other *Synemon* species (DSE 2003; Douglas 2007; Department of the Environment 2022). At least five other *Synemon* species occur within Victoria, which are either Threatened species in urgent need of direct management or remain too poorly known for their status to be assessed at all (DSE 2003; Douglas 2007).

Given the abundance of *S. plana* populations within Victoria, including within multiple reserves targeted specifically toward the conservation of this species, ongoing management will involve optimising ecological maintenance within existing sites, as well as extending conservation efforts into additional areas. The suitability of the intensive biomass management regimes required, which include grazing, prescribed burning and slashing, will vary between sites, based upon their broader conservation objectives. There has been an increased recognition and reappraisal of the role that sheep grazing land can play within biodiversity conservation, within south-eastern Australia (Dorrough *et al.*, 2007; New 2019). Low intensity agricultural land use can support a range of native Victorian grassland species, while simultaneously delivering production outcomes (Dorrough *et al.*, 2007). Untreated "native" pasture makes up approximately 22% of the agricultural estate (Mitchell *et al.*, 2019). In the absence of pasture enrichment, low intensity sheep farming can produce an intermediate level of disturbance within grassland landscapes that is amenable to *S. plana* conservation (McIntyre *et al.*, 2022). Accordingly, conservation policy should seek to support and encourage this style of land use, in order to incentivise its broader adoption by landowners. Currently, a prevalent hurdle to the retention of *S. plana* populations within agricultural landscapes is a tendency towards overly prescriptive regulation, that fails to recognise the heterogeneity of such landscapes, the resilience of *S. plana* to certain disturbance processes and the need for rapid and adaptive management responses to changing conditions, including weed incursion and spread. There is a pressing need for robust, quantified study of the impacts of different intensities of fire and grazing, upon *S. plana*. *S. plana* conservation initiatives, both within existing reserves and upon private property, must remain responsive to the ongoing impacts of anthropogenic climate change. Climate change holds the potential to disrupt key ecological balances within temperate grassland sites in certain regions of Victoria, such as c3 grass abundance and the vigour of woody vegetation encroachment into grassland landscapes. These changes will likely impose significant impacts upon some *S. plana* populations.

Within a Victorian context, targeted management activities that are specifically optimised towards *S. plana* conservation will not be appropriate for every site where the species occurs, and can potentially impose unwanted impacts to rarer and more sensitive biodiversity. Outside of existing reserves, a purist, single-species approach to *S. plana* conservation will remain most appropriate at the geographic extremes of *S. plana*'s distribution, where populations may represent unique genetic haplotypes. *S. plana*, like all native invertebrate species within Victoria (New 2019), is overdue inter-population genetic research, which will assist managers to assign future conservation priorities across its range (Clarke & O'Dwyer 2000; DAWE 2021a). The prominence that *S. plana* holds within Victorian conservation will continue to present funding opportunities for refined and targeted approaches to its conservation needs. Such opportunities will best be utilised in situations where they can provide robust benefits to other significant native biodiversity.

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Appendix A. Evolutionary Origins of *Synemon plana*

The Castniidae are believed to have originated within Gondwana (Hall and Holloway 1998, *cited in* Douglas 2004). The subfamily Castniinae comprises approximately 81 species in 30 genera across Central and South America and 44 species in mainland Australia. An estimated six Australian species are still currently undescribed (Edwards 2004 & Lamas 1995, *cited in* Douglas 2004; Kallies *et al.*, 2016; Kallies & Edwards 2018; Kallies *et al.*, 2020; Herbison-Evans & Crossley 2022). All the Australian Castniid species belong to genus *Synemon* (Doubleday 1846, *cited in* Douglas 2004).

S. plana is believed to have originated in central Australia, before gradually colonising Victoria (Edwards *cited in* Clarke & Whyte 2003). As *S. plana* continued expanding eastwards it appears to have encountered a hard topographical barrier in the form of the higher altitudes of the Great Dividing Range (Clarke & Dear 1998). *S. plana* then appears to have expanded northward into New South Wales (NSW) and the Australian Capital Territory (ACT) from a small founding population (Edwards, *cited in* Clarke & Whyte 2003). Currently, the timing of this event is unknown, although it is probably ancient, even when considered at a geological timescale (Edwards, *cited in* Clarke & Whyte 2003). The reduced dispersal ability of the adult female implies the presence of continuous grassland habitat across areas of northern Victoria, ACT and NSW, during the establishment of these new populations (Edwards 1991, *cited in* O'Dwyer & Attiwill 2000). *S. plana*'s reduced dispersal ability and reliance upon continuous habitat entails that the geographic distance between extant populations corresponds to their genetic relatedness (ACT Government 2020a). The larger population within Victoria has, for example, since become isolated from these northerly populations and has accordingly evolved its own unique genetic haplotype (Clarke & Whyte 2003).

Appendix B. Native Predators of *Synemon plana*

The characteristic mass, synchronised emergence of *S. plana* likely maximises recruitment, by sacrificing surplus individuals to predators, while the strongest moths survive to breed. The Willy Wagtail (*Rhiphidura leucophrys*) is a particularly prevalent predator of adult moths (Bainbridge & Longmore 2016). The moths are also targeted by Magpie Larks (*Grallina cyanoleuca*), Welcome Swallows (*Hirundo neoxena*) (DAWE 2020, *cited in* DAWE 2021a; Douglas 2004) and Australian Magpies (*Gymnorhina tibicen*) (B. Tomkins 2022, FoMP, *pers coms*.). The Striped Legless Lizard (*Delma impar*) is expected to prey upon *S. plana* as well (Kutt *et al.*, 1998; Australian Government 2009). Emerging adults are hunted by robber flies (Asilidae) (Clarke & O' Dwyer 2000) and dragonflies (Epiprocta) (MCMC 2020). Robber flies often become increasingly active during *S. plana* emergence events, which often indicates suitable conditions at a site for *S. plana* flight (The author 2022, *pers. obs.*). Spiders, including Araneidae (Orb-weavers) and Lycosidae (Wolf spiders) prey upon adult *S. plana* as well. Of these, Wolf Spiders are particularly prevalent predators, and they target freshly emerged, gravid females, which likely affects population numbers at certain sites (Douglas 2004).

Appendix C. Vegetation assessment

A vegetation survey of *S. plana* habitat should include data on species composition, dominance, and height (Kutt *et al.*, 2016). While there are a range of different approaches to gather this data, the specific floral survey approach that is adopted at a given site needs to be quick, simple, replicable and quantitative (Gibbons *et al.*, 2009). Surveying during the spring is ideal, because most floral species will be active and easiest to identify (Parsons Brinckerhoff Australia 2008). Photos should be taken in a standardised manner for any given approach, with attention paid to ensure that the height, direction and angle of the collected images is consistent (Parsons Brinckerhoff Australia 2008; Bainbridge & Longmore 2015). Accordingly, aerial drone photography is increasingly incorporated into modern approaches (Franceschini *et al.*, 2022).

Depending on the goals for the site and its specific monitoring objectives, it may be appropriate to maintain multiple permanent vegetation monitoring transects in order to generate representative data of the vegetation at different locations, from year to year. This type of monitoring would require that each transect receive its own vegetation assessment, on every monitored *S. plana* flight season (Kutt *et al.*, 2016). Transects should be assigned in a randomised manner, at an appropriate level of replicability to allow robust statistical analysis if needed (typically a bare minimum of 10 replicates). Vegetation data should be recorded at regular intervals along a transect depending upon its length (e.g., 40 points, spaced every 2.5 m along a 100 m transect (Kutt *et al.*, 2016). The species present at each intersecting point, and its height, can then be recorded (Kutt *et al.* 2016). The "Step Point Method" is a simpler approach for sites that do not require such a robust level of data (Sharp *et al.*, 2005; Parsons Brinckerhoff Australia 2008). In this approach, two transects are deployed along the longest axis of the site and the surveyor then records the plant species present at multiple point intersects along each transect, by taking 100 steps, and checking each step as a separate point intersect. Parsons Brinkerhoff Australia (2008) also recorded rocks, leaf litter and cryptogams as part of the data derived from this method and they then used this data to inform site management for an offset in Canberra, according to specific target thresholds for ground cover, species composition and tussock height. The Wheel Point Method (WPM) (Tidmarsh & Havenga 1955) has also been adopted in an Australian grassland context (Arzani & King 1994). In the WPM, a wheel is rolled along the transect that clicks at each desired interval (e.g., 1 m), which designates each monitoring point (Tidmarsh & Havenga 1955). This method can be simulated by handheld pedometers, allowing greater ease of use (Arzani & King 1994).

Species composition/richness can be derived by placing 1 m^2 quadrats at allocated spots, and then assigning percentage cover abundance scores for each species present, according to the Braun-

Blanquet method (1921, see van der Maarel 1975; Wikum & Shanholtzer 1978). In the Australian Capital Territory and New South Wales, the floral species that occur in grasslands have been assigned "significance scores", allowing for the floristic community present at *S. plana* sites to be numerically quantified for how well it represents the template values of its Endangered Ecological Community (Rehwinkel 2015; SMEC 2018). This approach is currently being adapted for use in Victoria's Temperate Grasslands of the South-eastern Highlands (Rehwinkel 2015). More general quadrat-based percentage estimates for native grasses, introduced grasses, native forbs, introduced forbs and bare ground are also useful data in assigning site quality values for *S. plana* habitat (Kutt *et al.*, 2016). A single quadrat was used at the halfway point of each transect in Kutt *et al.*'s (2016) method. Parsons Brinckerhoff Australia (2008) produced baseline data of the optimal species richness and percentage cover for their study site by placing a 400 m^2 quadrat in the centre of the best quality *S. plana* habitat. Bainbridge & Longmore (2015) contrasted similar quadrat flora surveys of core *S. plana* habitat (at 100 m^2) with quadrats placed in areas of the site that had previously shown no *S. plana* activity. Their dataset also included aspect and slope for each quadrat and recorded the presence of logs and branches on the site (Bainbridge & Longmore 2015).

The Golf Ball Method, developed at the Morgan Plant Ecology Lab at La Trobe University, in association with Parks Victoria (Schultz & Morgan 2010, *cited in* Schultz *et al.* 2017) has become one of the most widely adopted methods of assessing a grassland's biomass levels in Victoria, including in conjunction with *S. plana* management (Schultz *et al.*, 2017). This approach involves dropping 18 orange or pink golf balls into a 1 m^2 quadrat, and then photographing it from eye height. Each golf ball is then assigned a value of either 1 (>90% of the ball is visible), 0.5 (33–90% of the ball is visible) or 0 (<33% of the ball is visible), based upon how obscured by biomass it is (Schultz *et al.*, 2017). The total score of the golf balls then provides a metric to assign a biomass value to that region of the site (0–4: high biomass, 5–13: medium biomass, 14–18: low biomass) (Griffith & Nano 2011; Kutt *et al.*, 2016; Schultz *et al.*, 2017). Kutt *et al.* (2016) deployed two golf ball quadrats per flora survey transect, in order to generate this data.

Appendix D. Observing and recording *Synemon plana*

D.1. Standard methodology

S. plana can potentially occur in any grassland or grassy woodland within its known distribution, provided that it has access to suitable food plants (Enderby & Koehler 2006). In essence, *S. plana* surveys represent standardised attempts to observe and count flying males while also recording incidental observations of females on the ground (Kutt *et al.*, 2015). Robust identification of both sexes of adult *S. plana* is relatively straightforward, due to their characteristic behaviours and distinctive appearances. This identification simply requires that field staff have received some introductory training from somebody experienced with the species (Richter *et al.*, 2009) and are carefully instructed in the survey technique (Bainbridge & North 2007).

Ideally, *S. plana* surveys should be completed during the peak flying period (November to January) (DEWHA 2009). Surveying should not begin in temperatures under 20°C and should be completed during the warmest part of the day, between 10.00 hours and 14.00 hours, and under minimal wind and cloud cover (Gibson & New 2007; Greenville *et al.*, 2012; Kutt *et al.*, 2015). An *S. plana* survey should be accompanied by reference data from a known *S. plana* population, within close geographic proximity (Gilmore *et al.*, 2008), such as can be retrieved from the Ecological Consultants Association of Australia’s (ECA) GSM Flight Diary (https://ecavic.org.au/resources/gsm/) (Bainbridge & Longmore 2015). *S. plana* surveys should ideally be completed by pairs of surveyors, each working together closely, in order to maximise chances of identifying insects on the wing at a distance (Kutt *et al.*, 2015). Bainbridge & North (2007) suggest that increasing this to a team of four is better still; comprised of one leader/navigator, one counter, and two scribes, to record moth numbers and habitat notes separately. Cloud cover, wind speed and temperature should be recorded every half hour (Kutt *et al.*, 2015). Bainbridge and Longmore (2015) prioritised days for *S. plana* surveys with peak forecast temperatures of at least 25°C and Brown *et al.* (2012) extended their surveying to 15.00 hours, though continued to survey a site later into the afternoon, provided that male *S. plana* were still flying after 15.00 hours (Brown & Tolsma 2010). Richter *et al.* conversely extended their survey times through to 16.00 hours (2013a).

Fixed-point counts are most appropriate for small sites, and they allow surveyors to quickly monitor the presence or absence of *S. plana*, by counting flying males from a designated location, generally on the edge of the area of activity, for which global positioning system (GPS) data has been recorded (NSW Government Department of Environment and Conservation (DEC) 2007, *cited in* DEWHA 2009). Using a hand counter and stop-watch, the surveyor slowly turns in a 360° circle, over a thirty second interval, counting all flying male *S. plana* within 25 m, and remaining at the spot for at least six minutes, while taking care not to recount individuals (DEC 2007, *cited in* DEWHA 2009; Richter *et al.*, 2013a; Bainbridge & Longmore 2015). If a given patch of *S. plana* habitat proves too large to survey from a single point, then it may be appropriate to assign several survey points, and alternate between them, when completing the daily survey (DEC 2007, *cited in* DEWHA 2009). Generally, four daily replicates of a site survey are required per flight season, in order to ensure that a reasonably strong snapshot of *S. plana* abundance is captured (Kutt *et al.*, 2015). If no flying males are observed after spot counts, on four different visits, during optimal weather conditions, *S. plana* can be considered absent from the site (Richter *et al.*, 2013a) when under the typical constraints of surveying under development pressure (New 2012; New 2018). Within a reserve or offset where *S. plana* is monitored across multiple years, it is important that each count follows the same standardised timing schedule (e.g., three minutes spent counting, then replicated five times, with 30 second intervals between counts) (DEC 2007, *cited in* DEWHA 2009).

Though more time-intensive, transect surveys are advantageous because they allow surveyors to cover the entire site and to flush resting *S. plana* from vegetation while walking (Gilmore *et al.*, 2008; DEWHA 2009). The personnel completing a transect survey should attempt to move together at a steady pace, while remaining attentive for both flying males, and for displaying females (*Kutt et al.,* 2015). High numbers of recorded males will prevent assigning detailed individual data points for each recorded individual and accordingly, this will necessitate generalising GPS data across 100 m sections of the transect, with both the beginning and the end of the mass emergence recorded as GPS points, along with the time that they were entered (Kutt *et al.*, 2015). Bainbridge and North (2007) advise assigning GPS points after each five-minute interval spent walking along a transect, when encountering particularly high numbers of flying moths, and then entering observed numbers of moths for each point.

The single most important factor for a *S. plana* presence/absence survey is whether *S. plana* are flying at known sites nearby. This information provides a baseline reference to contextualise the survey results for the new area (Clarke 1999, *cited in* Gilmore *et al.*, 2008). The specific method used for a presence/absence survey for *S. plana* should be informed by the size and topography of the site (Brown *et al.*, 2012). A grassland can be surveyed with multiple parallel transects spaced between 5 and 50 m apart, as appropriate, while a linear roadside will often only allow for a single transect on either side of the road (Gison *et al.*, 2008; Brown & Tolsma 2010; Brown *et al.*, 2012). In some cases, site conditions can permit surveying from a vehicle, while moving at 10 km/hr (Brown & Tolsma 2010), such as within large, low-quality high-biomass sites, where *S. plana*, if present at all, tend to congregate near firebreaks and tracks (Rowell 2013a). When completing a vehicle-based *S. plana* survey, it is still necessary to stop once every 100 metres and perform a spot count of appropriate patches of habitat on foot (Rowell 2013a). Such an approach is of course suboptimal due to the increased potential of mortality to female *S. plana* and should be restricted to highly degraded areas with a low likelihood of detection, when survey resources are too sparse to allow for walking transects.

If *S. plana* is not detected on the first survey, subsequent visits should narrow the transect spacing to 25 m or less (Brown & Tolsma 2010). Historically, presence/absence surveys for *S. plana* have required that three site-wide surveys be completed within a single flight season. If three consecutive surveys fail to detect flying males under ideal conditions, this is considered indicative of *S. plana*' s absence at that site (Gilmore *et al.*, 2008), which, typically, is then legally sufficient to justify its development (New 2012; New 2018). While days of optimal weather and temperature should be prioritised, suitably detailed surveys completed on cool, cloudy or windy days can still provide adequate data for the presence/absence of *S. plana* at a site (Gilmore & Mueck 2012). Brown & Tolsma (2009) only listed a site as having an *S. plana* population if they were able to make five or more observations of individual *S. plana* there across the flight season. If *S. plana* is recorded outside of its known habitat range in Victoria, or within a particularly data-sparse area, ideally, a specimen should be collected by an appropriately permitted surveyor and provided to Museums Victoria (Gilmore *et al.*, 2008).

The data collected from a *S. plana* survey should be retained by the site manager, and then built upon with future surveys over subsequent years (Douglas 2004), which can then inform future ecological management plans for the site (DAWE 2021). High priority sites, such as Salisbury Bushland Reserve in the Wimmera, may require annual surveys to detect any changes in the resident *S. plana* population (Douglas 2004). The intensity of survey effort deployed at a given *S. plana* grassland should be determined by the site priorities and available resources for that area (E. Stone, HCC, 2022, *pers coms.*). On large sites, for example, *S. plana* are unlikely to emerge simultaneously, with adult *S. plana* progressively emerging from successive habitat patches according to specific changing microclimate factors, which can make standard transect surveys inefficient and inaccurate (Braby & Dunford 2006; DEWHA 2009; Kutt *et al.*, 2015). Sites that are repeatedly surveyed for *S. plana* from year to year will therefore generate more robust data, if subsequent surveys follow the same transects as those in previous years, according to a set order (Bainbridge & North 2007; Kutt *et al.*, 2015). It may be more appropriate, at particularly small sites, or sites where *S. plana* is known to be concentrated into a very small location, to monitor them using fixed-point counts, rather than transects (Bainbridge & North 2007; DEC 2007, *cited in* DEWHA 2009). Each daily replicate should ideally be separated by several days, in order to account for changes in emergence conditions (Biosis 2019b). Repeated, multi-year monitoring permits managers to detect spatial shifts in the population, which provides critically important information on the population health of *S. plana* at the site, which will subsequently inform the best management approaches to support it (Kutt *et al.*, 2015; Abzeco 2018).

D.2. Limitations of current methodology

Predicting the *S. plana* flight season each year is difficult. Flight season predictions often involve additional site visits before and after the core season has occurred, in order to gather the necessary data to determine the actual duration of the emergence (Gibson & New 2007). Standard survey techniques can only be engaged during optimal weather conditions, which causes severe time constraints (Richter *et al.* 2013a). Because the daily activity cycle of *S. plana* is so short, these constraints are compounded whenever there is a need to collect data on several populations simultaneously (Richter *et al.*, 2013a). This means that large numbers of personnel must be deployed simultaneously, if multiple sites need to be surveyed in the same area, such as is sometimes needed for reference data during presence/absence surveys (Richter *et al.* 2013a). Presence/absence surveying is rendered particularly unreliable by the mostly unquantified potential of *S. plana* larvae to extend their development across multiple seasons (New 2015). There are also issues associated with relying on integrated databases like the GSM Flight Diary for reference data (Bainbridge & Longmore 2015). Generally, these published records are derived from reference checks to determine the suitability of a given flight day and they are not robust or detailed enough to predict the days of peak *S. plana* emergence within the season, as is needed for abundance surveys (Bainbridge & Longmore 2015).

Abundance surveys are further limited by the tendency of *S. plana* to concentrate its core flight activity within a very narrow window of the day, even when optimal survey conditions present (DEWHA 2009; New *et al.*, 2007). This effectively invalidates abundance survey data collected in the morning or in the late afternoon (DEWHA 2009). Abundance data is drastically impacted by the El Niño/ La Niña cycle, and current approaches struggle to factor this into multi-year population survey data (Kutt *et al.*, 2015; New 2015). During the 2010–2011 season, for example, many *S. plana* surveys across Victoria were upset by wet, cloudy conditions, and a resultingly low number of adults was observed (Brown *et al.*, 2012). This interfered with many ongoing attempts to monitor local populations, producing disproportionately low numbers that were of no help in assessing *S. plana* population trends at the monitored sites (e.g., Sugarloaf Pipeline Alliance 2011; Jellie *et al.*, 2014). Conventional *S. plana* abundance survey approaches are also biased towards males, with transect surveys seldom exceeding an observation rate of one female per hundred observations (ACT Government 2017a). This produces an inaccurate picture of the site's total population size as well as preventing surveys from assessing the sex ratio of the population, which is important data relevant to a population's functional viability (Richter *et al.*, 2013a).

Despite a historic need for a centralised database for *S. plana* records within Victoria (New *et al.*, 2007; Richter *et al.*, 2009) and the subsequent rise of several online platforms that can assist with this (e.g., Victorian Biodiversity Atlas (VBA), INaturalist, GSM Flight Diary, Atlas of Living Australia), much ongoing monitoring of *S. plana* still fails to utilise these public data repositories, with the consequence that many observations remain informal anecdotes and are not integrated into public databases. Unless a specific research permit has been assigned by DELWP, there is no legal requirement for *S. plana* survey data to be entered into the VBA (S. Hadden, 2022, *pers coms.*). Additionally, client confidentiality can restrict consultants from making their *S. plana* survey results public (Bainbridge & Longmore 2015). The Parliament of Victoria's 2021 inquiry into biodiversity declines in Victoria identified problems with data collection and effective monitoring, stemming from a lack of central coordination, funding limitations, over-reliance on citizen science and ease-of-use issues with available databases. Several local councils (e.g., Nilumbik Shire Council, Kingston City Council, Macedon Ranges Shire Council, *cited in* Parliament of Victoria 2021) submitted concerns about the VBA's capacity to provide the area-specific data that they needed in order to assign ecological management priorities, and a lack of resources to gather this data themselves. DELWP has recently launched the VBA Go mobile application, seeking to address such concerns (Parliament of

Victoria 2021).

D.2.1. Cross-identification problems

Robust identification of adult *S. plana* of both sexes is relatively straightforward, due to their characteristic behaviours and distinctive appearances, provided field staff receive some introductory training from somebody experienced with the species (Richter *et al*., 2009). Staff that are new to invertebrates may nevertheless falsely assign a *S. plana* identification to a range of similarly sized grassland insects (e.g., Yellow-winged Grasshopper (*Gastrimargus musicus*), Chequered Copper Butterfly (*Lucia limbaria*), Meadow Argus Butterfly (*Junonia villida*)), particularly when the animal is only seen at a distance (The author 2022, *pers obs*.). This can be particularly difficult to avoid on widely spaced transects. Consequently, in situations where low numbers of isolated individuals are recorded by inexperienced staff, such records should be treated with care, unless they can be verified or replicated by subsequent observations (The author 2022, *pers obs*.).

New *et al.* (2013) note that the habits, lifecycles and distributions of the other *Synemon* species present in Victoria (e.g., Striated Sun Moth (*S. collecta*)) are very poorly known (see Douglas 2004). Under changing climate conditions, such species might plausibly coincide with *S. plana* habitat in novel ways that could complicate surveys (New *et al*., 2013). *S. collecta* is particularly notable as it also favours *Rytidosperma*-dominated grasslands and emerges from December to mid-January (Douglas 2004). Older and more battered looking *Synemon* moths are particularly difficult to identify with confidence due to their variation from typical colour and patterning in the species (B. Tomkins 2022, *pers coms*.). The darker colouration of freshly emerged adult *S. plana*, can also confuse observers, particularly at the onset of the season (B. Tomkins, FoMP, 2022, *pers coms*.).

D.3. Emerging approaches

Improving understanding of the food plants utilised by *S. plana*, and its specific habitat and microhabitat preferences, will allow greater levels of precision in presence/absence surveys, and better accuracy in abundance surveys (Bainbridge & Longmore 2015; Kutt *et al*., 2016). Percentage cover of *Rytidosperma* and *Austrostipa* have so far proven to be reliable predictors of *S. plana* presence upon a site, with increasing biomass shown to act as a generally robust negative indicator of *S. plana* presence, after a certain point (Kutt *et al*., 2016; A. Kutt 2022, *pers coms*.). These and other factors, such as aspect and landscape position can be identified relatively simply through aerial photographs or remote sensing and then mapped with species distribution models (Lahoz-Monfort *et al*., *cited in* Kutt *et al.,* 2015). Mapping and then surveying the resulting areas of ideal habitat, will therefore lead to improved knowledge of localised *S. plana* distribution and abundance and allow more efficient use of funding and survey time (Kutt *et al*., 2015). There is also capacity to produce time-to-detection (TTD) models (see Henry *et al*., 2020; Strebel *et al*., 2021) from *S. plana* data, particularly for sites that have been routinely surveyed for several years. A TTD modelled abundance survey would be able to generate abundance data for a population, based upon the speed at which *S.*

plana is observed at a site in optimal conditions, which would, in turn, reduce the need for repeated survey days (Kutt *et al.*, 2015).

Pupal case surveys have been widely recommended as a supplementary survey technique for *S. plana* (Australian Government 2009), that allows populations to be monitored independently of weather conditions, time of day or general day-to-day fluctuation in adult emergence rates (Richter *et al.*, 2013a). This approach can provide valuable information for the conservation management of *S. plana*, including the sex ratio of a population, and a more robust estimate of its total size (Richter *et al.*, 2013a; Richter *et al.*, 2013b), although it requires a high level of technical expertise, because the pupal stage of *S. plana* has not been formally described (New 2012; S. Hnatiuk, FoG, 2022, *pers coms.*). *S. plana* pupal casings remain intact under natural conditions for a period of at least three weeks, and they are durable to the effects of high temperature, rainfall and trampling by kangaroos (Richter *et al.*, 2013a). It is necessary to either remove the casings on each count, or to mark them with paint, in order not to overestimate *S. plana* emergence by re-counting individuals (Richter *et al.*, 2013a). In 2013, pupal case identification was effectively taught to citizen scientists in Canberra and deployed as part of broadscale population monitoring of *S. plana* (Richter *et al.*, 2013a). This study surveyed grasslands by assigning 12 randomised 1 m^2 quadrats across the site for analysis (Richter *et al.*, 2013a). Each quadrat was visited four times across the flight season, with at least a week between each visit (Richter *et al.*, 2013a). *S. plana* cases were removed on each visit, generally with their surrounding soil and silk webbing attached, in order to avoid damaging them (Richter *et al.*, 2013b). The samples were then cleaned, identified and sexed under a microscope, through comparison with reference specimens, achieving an accuracy level exceeding 98% (Richter *et al.*, 2009; Richter *et al.*, 2013a; S. Hnatiuk 2022, *pers coms.*). Pupal surveys have also been deployed as supplementary survey approaches by SMEC, at York Park, Canberra (ACT Government 2017c) and by the Sugarloaf Pipeline Alliance (2011) at Sheoak, Yea, though they required an external entomologist to examine the samples. *S. plana* pupal case surveying is seldom used, due to the specialist skillset required. Early monitoring projects in the Australian Capital Territory (ACT), that were spearheaded by Anett Richter, have since abandoned this technique due to a lack of available technical expertise (S. Hnatiuk 2022, *pers coms.*). Additionally, pupal case surveys are less likely to produce robust results when deployed in high biomass areas, where it is generally much more difficult for surveyors to find the cases (Richter *et al.*, 2013b). The level of detail required in pupal case surveying makes this technique suitable to use in conjunction with more structured searches for female *S. plana*, which are normally too time-intensive to survey for in a robust manner (ACT Government 2017c). Abundance surveys can also incorporate pupal case searches in more of an *ad hoc* manner, when unsuitable weather has prevented the completion of the necessary four site visits, allowing the surveyor to bridge the gaps in their standard approach at the end of the season (Rowell 2013a; Richter *et al.*, 2013a).

Mark Release Recapture (MRR) of captured adult *S. plana*, similarly holds potential as a supplementary long-term monitoring technique for very small sites of particular importance (Richter *et al.*, 2013a). MRR presents the most robust method of gauging a population's abundance, with no chance of re-counting individuals (as within standard surveys) (Richter *et al.*, 2013a). MRR is extremely time-consuming for *S. plana*, because the short adult life of the species ($\bar{x}$ = 1.08 days), necessitates confining such surveys to single days (Richter *et al.*, 2013a). MRR may however allow for more detailed assessment of moth survival during the flight day (Richter *et al.*, 2013a). One such threat to adult *S. plana* survivorship at small urban sites is avian predation (Bainbridge & Longmore 2016) (e.g., Appendix E4).

Surveying the soil of a grassland for the presence of *S. plana* larvae is time consuming and destructive (ACT Government 2017c) and demands specialist identification skills (New 2012), due to the fact that the larval stages of *S. plana* have not been formally described (ACT Government 2017a). However, in situations in which other activities must be performed on a site that will cause soil disturbance (e.g., pitfall trapping for lizards or small mammals), then larval surveys may present a viable method to procure supplementary population data about *S. plana*, in terms of population density, the age cohorts present and the food species that they are using (DEWHA 2009; ACT Government 2017c). Soil surveys for *S. plana* larvae have been completed at ACT sites, although this approach remains in a preliminary stage, as a survey technique (SMEC 2015, *cited in* ACT Government 2017c).

Appendix E. Additional case studies

E1: Craigieburn Grassland Reserve/Galgi Ngarrk

Craigieburn Grassland Reserve is habitat to one of the most significant *S. plana* populations in Victoria but has suffered from the absence of a formal surveying programme and a lack of dialogue between stakeholders. Craigieburn Grassland Reserve is a 400 Ha conservation reserve in northern Melbourne, comprising Temperate Native Grassland of the Victorian Volcanic Plains, in association with wetland areas and riparian habitat (DNRE 1998; Douglas 2004; MCMC 2021). In 2009, Craigieburn Grassland was listed as one of Melbourne's most significant parks for biodiversity conservation (DSE). In addition to *S. plana*, Craigieburn Grassland retains populations of *Delmar impar, Dianella amoena*, Plump Swamp Wallaby Grass (*Amphibromus pithogastrus*) and one of the only remnant populations of Curly Sedge (*Carex tasmanica*) in Melbourne (DSE 2009). The site was also one of the last locations in Victoria where the Grassland Earless Dragon was recorded (Beardsell 1997, *cited in* DNRE 1998). Accordingly, Craigieburn Grassland is listed as a Nationally Significant ecological community (DNRE 1998). The site's high conservation values derive in part from its large size, varied topography and connectedness, through riparian corridors, with other ecological areas in north Melbourne (Beardsell 1997, *cited by* DNRE 1998; Bainbridge & North 2007).

In pre-European times, this grassland landscape was shaped, over a period of thousands of years, by cultural burning deployed by the Wurundjeri people (DNRE 1998). MCMC (2021) engagement with Wurundjeri elders has assigned the Woiworrung language name "Galgi Ngarrk", to this site, which means "Back Bone" (Parks Victoria 2013). Galgi Ngarrk is connected to Bababi Marning (Cooper Street Grassland) through a series of riparian parklands, collectively named "Marran Baba", or "Body of Mother", although these names are not yet formally accepted by the Victorian Government (MCMC 2021). Following European settlement, this landscape was grazed by cattle and sheep, and this continued until the early 2000's (T. Liddel 2022, *pers coms*). Historic burning of these grasslands ceased entirely when it became ranch land (DNRE 1998). During this time, portions of Craigieburn Grassland were also treated with superphosphate and sown with exotic pasture grasses (DNRE 1998). The site's rocky topography appears to have kept grazing levels to a low-density regime, which has allowed pockets of high biodiversity to survive within the landscape (DNRE 1998). When Craigieburn Grassland Reserve was established, all stock were removed, which resulted in an influx of weeds throughout the site (DNRE 1998). Freeway development to the north and east has limited connectivity to Craigieburn Grassland, which is particularly significant for fauna dispersal (Ecology Australia 1996, *cited in* MCMC 2021). This has contributed to gene flow issues for *S. plana* (van Praagh 2004). Craigieburn Grassland's resident Eastern Grey Kangaroo mob (BNRE 1998; Bainbridge & North 2007) appears to have increased in numbers and density over recent years (The Author, *pers obs.*).

S. plana was discovered at Craigieburn Grassland in 2004, when the Craigieburn Bypass Development proposal (van Praagh 2004) spurred MCMC to survey the site for threatened species (Bainbridge &

North 2007). Surveys established *S. plana* to be almost ubiquitous across the reserve, except for areas that had historically been treated with superphosphate (Bainbridge & North 2007). During the mid-2000’s, more than a thousand *S. plana* were observed across a single flight season (Gibson 2006). *S. plana* is particularly concentrated around rock escarpments and low rises and occurs in particularly high numbers in the saddles between rock knolls (Bainbridge & North 2007). Conversely, *S. plana* is comparatively absent within low-lying areas of dense biomass, such as those dominated by *T. triandra* and Common Tussock-grass (*Poa labillardierei*) (Bainbridge & North 2007). Annual total records of *S. plana* vary heavily from year to year, despite comparable survey conditions (Enderby & Koehler 2006). This has led researchers to infer the presence of at least two temporally and reproductively isolated *S. plana* populations, which emerge separately on sequential years (Enderby & Koehler 2006). Craigieburn’s *S. plana* population is important to the wider conservation of the species, due to the large size and protected status of its grassland habitat (New *et al.*, 2007). Craigieburn Grassland has previously been highlighted as suitable for large-scale research projects, to develop quantified ecological management techniques for *S. plana* conservation (New *et al.*, 2007). The reserve, and its resident *S. plana* population, continues to be used as an important reference site for the broader landscape around Epping (e.g., Biosis 2008; Brett Lane & Associates 2018).

MCMC and Parks Victoria began managing this *S. plana* population during the mid-2000’s (New *et al.*, 2007). Bainbridge & North (2007) advised more detailed vegetation surveys and topographic mapping of Craigieburn Grassland as a follow-up to their surveys in order to gauge the habitat preferences of *S. plana* across the grassland, with greater precision (Bainbridge & North 2007). Parks Victoria currently performs woody weed control at Craigieburn Grassland, targeting gorse and Sweet Briar (*Rosa rubiginosa*) and manage biomass through ecological burns, deployed across a 3–4 year cycle (T. Liddel*,* 2022, *pers coms.*). Burns are timed to coincide with targeted weed control, in order to maximise outcomes from limited site budgets (T. Liddel 2022, *pers coms.*) and all burns are strategically deployed in order to avoid rock knolls and other areas of high biodiversity (T. Liddel 2022, *pers coms.*). *N. neesiana* is a prevalent weed at Craigieburn Grassland, as are *N. trichotoma* and Artichoke Thistle (*Cynara cardunculus*) (DNRE 1998; New *et al.*, 2007). Parks Victoria identify major funding constraints as a continuing obstacle to their stewardship of the site (T. Liddel, 2022, *pers coms.*). The organisation holds no current records of *S. plana* and, in recent years, they have ceased managing the site for the specific requirements of *S. plana* (T. Liddel 2022, *pers coms.*). *S. plana* conservation will now again be incorporated within a new annual works plan, in development for Craigieburn Grassland (T. Liddel 2022, *pers coms.*). The Friends of Merri Creek have planted 7,500 *Rytidosperma* within Craigieburn Grassland Reserve, which were specifically chosen to bolster the site’s resident *S. plana* population (MCMC 2021). Ultimately, failure to follow-up on initial surveying of this population has led to a lack of robust data on the current site occupancy of one of the most important *S. plana* populations in Victoria. This has resulted from a lack of communication between stakeholders and a general lack of funding for a significant reserve with high biodiversity values.

E2. Broadmeadows Valley Park

Broadmeadows Valley Park (BVP) demonstrates the resilience of *S. plana* to specific forms of habitat disturbance. BVP forms part of a large, interconnected corridor of native vegetation and suburban parklands between roadways and areas of urban development (Ecology Australia 2020). Much of the landscape of Broadmeadows and Westmeadows was initially used for grazing during the 1840’s (Melbourne History Research Group 2018). During the 1950’s large areas of Broadmeadows were rapidly developed by the Housing Commission of Victoria (Melbourne History Research Group 2018).

Conservation management of this undeveloped area began in the mid-1990’s when the area become a part of the newly formed City of Hume, formalising its use as a public recreational space (Melbourne History Research Group 2018; D. Pascuzzo 2022, *pers coms*). The need to mitigate wildfire risk and manage the area for public amenities necessitated regular mowing of open grassy areas across the park, which were mown according to a six-week cycle (D. Pascuzzo 2022, *pers coms.*). BVP’s open sward grassland areas are partly *Rytidosperma* dominated, but *S. plana* is particularly prevalent throughout modified areas that are dominated by *N. neesiana* (E. Stone 2022, *pers coms*), where it occupies around 90 Ha of habitat.

While HCC had previously observed high numbers of *S. plana* within BVP, the population was first formally documented in 2015, during a general fauna survey (Biosis 2017D). HCC staff observed very high numbers of *S. plana* flying throughout the mown areas of the parkland habitat, identifying that the site retained one of the largest populations of *S. plana* in greater Melbourne (Biosis 2017D; Pascuzzo 2022, *pers coms*), and engaged external consultants to map its habitat occupancy (Ecology Australia 2020). A follow-up survey in the 2016/2017 flight season, surveyed 70% of BVP, and recorded 3719 individuals (D. Pascuzzo 2022, *pers coms.*). Subsequent surveys identified that *S. plana* was particularly concentrated on north-west facing slopes (Ecology Australia 2020). BVP’s *S. plana* population has routinely been used as a reference population for *S. plana* surveys in north west Melbourne (Ecology & Heritage Partners 2021, J. Harris, Wildlife & Ecology 2022, *pers coms.*). The 2019/2020 *S. plana* survey also recorded seven incidental observations of Threatened Growling Grass Frogs (*Litoria raniformis*) occupying BVP’s creek line (Ecology Australia 2020).

S. plana appears to have proliferated throughout BVP’s extensive swards of mown *N. neesiana* (E. Stone 2022, *pers coms.*). However, managing biomass within BVP’s *S. plana* habitat requires attention to the *S. plana* emergence cycle, and by optimally timing slashing in November, HCC's mowing schedule can continue without impacting *S. plana* within the core flight season (Ecology Australia 2020). *S. plana* habitat at BVP is threatened by weeds, including *Phalaris aquatica*, Kikuyu Grass (*Pennisetum clandestinum*), Artichoke Thistle, Paterson’s Curse and *Nasella trichotoma;* all of which require control with spot-sprayed herbicides (Biosis 2017; Ecology Australia 2020).

Eric Stone, Conservation Team Leader (2022, *pers coms.*), works to maintain the biodiversity values of a few small areas of intact VVP grassland within the broader landscape. Given the prevalence of *S. plana* throughout highly modified areas, he sees no specific need to maintain the high-quality remnants as *S. plana* habitat (E. Stone 2022, *pers coms.*). These VVP remnants support threatened and locally rare flora, of conservation significance, including Sun Orchids (*Thelymitra* spp.), Grass Trigger Plants (*Stylidium graminifolium*) and one of the largest populations of *Geranium* "Species 1" in greater Melbourne, which could all be adversely impacted by *S. plana* focused management approaches (E. Stone 2022, *pers coms.*). BVP's *S. plana* population has proliferated due to intensively mowing a noxious weed. Under the right conditions, *S. plana* has high potential to occupy large areas that would normally be considered of low conservation value.

E3. Cooper Street Grassland/Bababi Marning

Cooper Street Grassland has a large urban *S. plana* population that benefits from the protected status of its habitat as a reserve, but nevertheless suffers novel edge effects from fragmentation. Melbourne's Cooper Street Grassland Reserve, in Campbellfield, supports a range of significant taxa, including the Endangered Growling Grass Frog (*Litoria raniformis*) (State of Victoria 2020) and the most intact patch of riparian Woolly Tea Tree (*Leptospermum lanigerum*) scrub in north eastern Melbourne (Beardsell 1997, *cited in* MCMC 2021). The site comprises 23 Ha of core grassland habitat; purchased by the Victorian Government in 1994, combined with an adjacent 15.5 Ha block; which was added as an outcome of the Craigieburn Bypass hearings, and various additional blocks of acquired farmland, which together total 52 Ha (Bainbridge & Longmore 2015). In 1990, this site was identified by the Victorian government as a high priority area of VVP, for acquisition as a conservation reserve (State of Victoria 2020), which was subsequently formalised as part of the Craigieburn Bypass Development hearings (van Praagh 2004; Bainbridge & Longmore 2015). MCMC engaged Wurundjeri elders to assign a name, in the Woiworrung language, to Cooper Street Grassland (MCMC 2021; Parks Victoria 2013). The site name "Bababi Marning", meaning "Mother's Hand", has not so far been formally adopted by the Victorian Government (Parks Victoria 2013; MCMC 2021).

S. plana was discovered at Cooper Street Grassland, in 2004, during surveys spurred by the Craigieburn Bypass Development proposal (van Praagh 2004; Bainbridge & Longmore 2015). *S. plana* occupies several square kilometres of grassland habitat in and around Cooper Street, including adjacent previously privately-owned properties, that had historically supported cattle and horse grazing (New, *et al.*, 2007; Bainbridge & Longmore 2015). In 2008, A total of 849 *S. plana* were counted during a survey, resulting in a total population estimate of 4000 *S. plana* for the site (Bainbridge & North 2010, *cited in* Bainbridge & Longmore 2010). After several years of comparatively low numbers of emerging moths (Bainbridge & Longmore 2015), another peak season was recorded in the summer of 2014/15, when an estimated 1198 moths were recorded. The destruction of 50 Ha of grassland habitat surrounding the reserve has resulted in the loss of several peripheral *S. plana* populations (Bainbridge & Longmore 2015). The surviving *S. plana* population is isolated from other suburban remnants due to the intensive landscape-level modification that has occurred across Melbourne's northern development fringe (Bainbridge & Longmore 2015).

S. plana favours low biomass areas of Cooper Street Grassland (Bainbridge & Longmore 2015). Biomass reduction is primarily achieved via a 3–5 year cycle of ecological mosaic burns, managed by

Parks Victoria (Parks Victoria 2013; Bainbridge & Longmore 2015). Bainbridge & Longmore (2015) consider this burn regime to have contributed to a general increase in *S. plana* numbers at the site. There are high numbers of kangaroos at Cooper Street Grassland, whose grazing intensifies after a burn, which is considered to have been helpful in maintaining a desirable site structure for *S. plana* (Bainbridge, *cited in* Bainbridge & Longmore 2015). There are currently no active monitoring programmes in place for *S. plana*, although Parks Victoria' s existing biomass reduction measures are considered sufficient to support the species (M. Longmore 2022, *pers coms.*; T. Liddel, Parks Victoria, 2022, *pers coms.*).

During *S. plana* surveys in the 2014–15 season, MCMC staff observed male moths congregating along the eastern urban boundary of the reserve (Bainbridge & Longmore 2016). Approximately 12% of the recorded male *S. plana* were distracted by sunlight glinting off yellow glass, which resulted in them concentrating their patrol flights around the disturbed site edge (Bainbridge & Longmore 2016). This interaction with glass rubbish has been proposed to reduce the reproductive output of the population, and expose the affected males to increased avian predation (Bainbridge & Longmore 2016). These authors (Bainbridge & Longmore 2016) advise an annual inspection for and clean-up of glass, prior to each flight season. MCMC (2021) continues to advocate for the addition of private pockets of grassland habitat with remnant *S. plana*, to the broader system of grassland reserves along the Merri Creek. Managing such patches specifically, to form low-biomass *Rytidosperma* swards, will support the area' s meta-population of *S. plana*, hopefully improving gene flow to *S. plana* at the site (van Praagh 2004; MCMC 2021). The establishment of Cooper Street Grassland Reserve has effectively preserved the core habitat of this *S. plana* meta-population, but the area requires ongoing targeted management in order to compensate for surrounding urban development and the loss of connectivity and disturbance that has resulted (Bainbridge & Longmore 2016).

E4. Amberfield Nature Reserve

Amberfield Nature Reserve demonstrates the complexities of managing a very small urban site for *S. plana* conservation. Amberfield Nature Reserve is a tiny, 1.8 Ha grassland reserve in Craigieburn, Melbourne, that is currently managed by HCC. It is a VVP temperate grassland (D. Pascuzzo 2022, *pers coms.*), that supports *Dianella amoena* and *S. plana* (Biosis 2008). Amberfield Nature Reserve appears to have once been farmland, although it is unclear if it was previously stocked (D. Pascuzzo 2022, *pers coms*). Interruptions to the fire cycle across this time, have likely contributed to a resulting reduction in floral diversity throughout the site (Biosis 2008). Amberfield Nature Reserve nevertheless retains a particularly high diversity of *Rytidosperma* and *Austrostipa* spp., which form a naturally sparse sward, growing on exposed, sedimentary soil (E. Stone 2022, *pers coms.*). The grassland is surrounded by urban development and an adjacent golf course, which provides additional habitat to the *S. plana* population (D. Pascuzzo 2022, *pers coms.*). The site entered HCC management in 2009 and a Trust for Nature Covenant was placed on the reserve in 2016 (Biosis 2008; Jacobs 2017; D. Pascuzzo 2022, *pers coms.*).

S. plana was first recorded at Amberfield Nature Reserve in 2005 (D. Pascuzzo 2022, *pers coms*). The *S. plana* population was annually surveyed by MCMC from 2010–2021, using Craigieburn Grassland as a reference site (Bainbridge & Longmore 2016; D. Pascuzzo 2022, *pers coms*). Although the *S. plana* population within the reserve itself appears to have a critically low population (Bainbridge & Longmore 2016), 100 moths were counted in the adjacent golf course in 2007 (D. Gilmore, Victorian Biodiversity Atlas). The small size of the reserve, coupled with its internal dividing fences, results in a disproportionately high rate of avian predation from Willy Wagtails, House Sparrows and Common Starlings, during the flight season (Bainbridge & Longmore 2016). During *S. plana* surveys in the 2014/2015 season, two of the four *S. plana* detected were eliminated by sallying Willy Wagtails (Bainbridge & Longmore 2016). Given the small size of the population, the surveyors were concerned that this predation posed a direct threat to its continued survival (Bainbridge & Longmore 2016). Attempts to control this by installing ‘holographic flash’ bird-scarer tape on the fence showed initial promise (Bainbridge & Longmore 2016). However, by the following season, resident Willy Wagtails were no longer deterred by the tape, and had resumed preying upon emerging *S. plana* (MCMC 2017). While survey work over the past decade has demonstrated a general increase in the *S. plana* population present within the reserve, its long-term viability is uncertain due to its small size, restricted habitat, limited gene pool, isolation from other populations and exposure to pronounced edge effects (E. Stone, M. Longmore 2022, *pers coms.*). HCC have since removed internal fencing from the reserve wherever expedient, and removed the wire from all remaining fencing, in order to reduce perching habitat (D. Pascuzzo, E. Stone 2022, *pers coms.*). When Amberfield Nature Reserve’s Operation Plan expires in 2024, HCC will likely cease annual monitoring of *S. plana*, in lieu of both the new “Vulnerable” Threat Status of *S. plana* in Victoria, and the proven effectiveness of current site management (D. Pascuzzo 2022, *pers coms.*). Eric Stone, the site’s current land manager (2022, *pers coms.*), considers biennial monitoring to be a more appropriate use of site budgets, which will allow for more attention towards other significant biodiversity.

HCC on-ground works are guided by an existing Operation Management Plan (D. Pascuzzo 2022, *pers coms.*). Initially, Sweet Briar was eliminated from the site through intensive cut-and-paint of mature individuals and spot spraying of germinants with herbicide (Biosis 2008). Spot spraying was also used to control a suite of weedy grasses and herbs, including Toowoomba Canary-grass, Cocksfoot, Perennial Ryegrass, Artichoke Thistle and Cape Weed (Biosis 2008). Biomass is controlled at Amberfield Nature Reserve primarily through ecological burning. While under an Offset Management Plan Amberfield Nature Reserve was required to be burnt across a two-year cycle, with 50% of the site burnt annually (D. Pascuzzo 2022, *pers coms.*). The burn regime strives to maintain an inter-tussock spacing of 10–50% of the ground cover (Biosis 2008). The site’s current management plan recommends burning during the autumn in order to avoid the *S. plana* flight season (*E. Stone* 2022, *pers coms.*). La Niña rains have resulted in pulses of weedy biomass in recent years, with the unburnt sections of the site rapidly accumulating weedy grasses and requiring costly manual brush cutting and biomass removal, as a supplementary measure (D. Pascuzzo 2022, *pers coms.*). Stone (2022, *pers coms.*) is concerned that biennial burning may caused adverse impacts to other biodiversity at the site over the long term, linking an observed decline in native Fabaceae as a potential consequence of the intensive burn cycle. He has accordingly sought to burn in a more adaptive and reactive manner, focussing upon smaller patches of habitat at a time (E. Stone 2022, *pers coms*.). Past assessment has recommended that *R. racemosa, R. setacea* and *R. caespitosa*, derived from locally sourced seed, be planted in areas where weeds have been eliminated (Biosis 2008). In 2015, MCMC staff observed Threatened Amethyst Hairstreak Butterflies (*Jalmenus icilius*) at the site, while surveying for *S. plana* (D. Pascuzzo 2022, *pers coms*.). HCC has subsequently elected to selectively restore *Acacia mearnsii* to areas of Amberfield Nature Reserve in the interim, in order to provide habitat for this species, whose needs are being balanced against optimal management for *S. plana* habitat (D. Pascuzzo, E. Stone 2022, *pers coms*.). The small size of Amberfield Nature Reserve and the low size of it's *S. plana* population exacerbate the impacts of multiple edge effects. Maintaining *S. plana* under such conditions is more expensive and requires much more targeted management approaches than are needed at larger sites.

E5. Mount Piper

The *S. plana* population at Mt Piper is one of Victoria's best monitored populations, though shares habitat with other threatened species of indeterminate status. This creates a direct obstacle to assigning site management priorities for this habitat. Mt Piper, Broadford, lies 80 km north of Melbourne, on the north-western edge of the Central Uplands Bioregion. The site consists of a single 456 m tall volcanic plug, with VVP grassy woodland on the lower slopes (*Figure* 11. Britton *et al*., 1995). The site is notable for providing habitat for a Nationally Significant community of Threatened lepidopterans, including the Large Ant Blue (*Acrodipsas brisbanesis*) and the Small Ant Blue (*A. myrmecophila*) Butterflies, in addition to one of the first populations of *S. plana* to be rediscovered in Victoria (Britton *et al*., 1995). The combined presence of three umbrella conservation species within the same site initially helped to elevate Mt Piper's prominence as a conservation area and encouraged engagement with the site by conservation volunteers and citizen scientists (New 1997).

Figure 11. Mt Piper, viewed from grassland habitat to the north-east (Photo credit: S. Sinclair, 2022).

The core of Mt Piper's large *S. plana* colony is located on the north-eastern face of the mountain (O'Dwyer & Attiwill 2000; DSE 2004) and is surrounded by cleared grazing properties (Britton *et al.*, 1995; Dear 1997). This *Rytidosperma eriantha* dominated grassland was likely created by forest clearing, during early European settlement and then maintained in this state by private sheep grazing (White *cited in* DSE 2004). *S. plana* is therefore likely to have been much less common at the site beforehand, likely spread diffusely throughout grassy woodland areas and moving from clearing to clearing after fire events. The original property owners did not improve the soil with fertilisers during their land use, which allowed *S. plana* to survive at the site (Douglas 2004). Mt Piper's Threatened Butterfly Community was formally listed in 1991, and the site then became a conservation reserve (DSE 2004). *S. plana* was first discovered at Mt Piper a year later (Britton *et al.*, 1995). During the early days of *S. plana*'s national rediscovery at various grassland sites across the ACT, NSW and Victoria, Mt Piper represented a critically important site for the species and was one of only a tiny handful of *S. plana* habitats known at the time (Dear 1997). In 1995, the north-eastern grassland was purchased by the Australian Nature Conservation Agency and added to Mt Piper Conservation Reserve (Douglas 2004). Stock were removed from the landscape following its purchase (Douglas 2004), and native grazing emerged as the site's primary means of biomass reduction (DSE 2004; T. FitzGerald 2022, *pers coms.*).

During the 1996/1997 flight season, Mt Piper's *S. plana* population was estimated to be between 300-700 individuals (Dear 1997). Parks Victoria, with the support of the Friends of Mt Piper (FoMP), monitored the population annually, using a grid of thirty permanent transects, until 2007 (T. FitzGerald 2022, Parks Victoria, *pers coms.*). Population monitoring was subsequently taken over by FoMP (B. Tomkins, 2022, *pers coms.*). Three similar-sized *S. plana* populations were discovered in nearby sheep paddocks, which were monitored in conjunction with the core population at Mt Piper (B. Tomkins 2022, *pers coms.*). FoMP assume these populations to retain occasional gene flow across the landscape (B. Tomkins 2022, *pers coms.*). Recent records show these three new *S. plana* populations to have remained relatively stable, despite fluctuating with the El Niño/La Niña cycle (B. Tomkins 2022, *pers coms.*). In contrast, the core *S. plana* grassland within Mt Piper Conservation Reserve appears to have declined due to insufficient site maintenance, and encroachment by Golden Wattle (*Acacia pycnantha*) (B. Tomkins 2022, *pers coms.*). FoMP collected monitoring data across most *S. plana* flight seasons until 2021 (B. Tomkins 2022, *pers coms.*). These records were sent to DELWP, intended for VBA submission (B. Tomkins 2022, *pers coms.*). In 2021, FoMP became aware that none of their data had been entered or made publicly available (B. Tomkins 2022, *pers coms*) due to limited resources within DELWP. FoMP are currently in the process of manually entering 15 years of accumulated *S. plana* site data into the VBA (K. Boehm 2022, *pers coms.*). The group has expressed frustration that monitoring and care of such a significant conservation area has largely been left to the goodwill of volunteers, and stresses a need for external funding in order to adequately monitor the reserve, perform restoration works, and complete the necessary data entry (B. Tomkins 2022, *pers coms.*).

The north-eastern grassland at Mt Piper is particularly prone to disturbance due to its proximity to neighbouring agricultural properties (Britton *et al.*, 1995). Historically, the primary threat to this population was invasion by weedy grasses, particularly *Lollium perenne* and *Holcus lanatus* (O'Dwyer & Attiwill 2000). After stock grazing ceased, these exotic grass species rapidly invaded from surrounding properties, swamping *Rytidosperma* populations within the site (O'Dwyer & Attiwill 2000). Cheryl O'Dwyer (Dear 1997) and Anne Jelinek (1991, *cited in* DSE 2004) advocated for the reintroduction of sheep grazing to Mt Piper's grassland, and consider the cessation of grazing to have been responsible for the influx of weedy grasses (Dear 1997). In 2007, Parks Victoria installed an electric fence along the grassland's northern boundary and then crash grazed the grassland using a sheep flock owned by an adjacent landholder (T. FitzGerald 2022, *pers coms.*). The grassland appears to have restabilised somewhat over recent years and has required little further biomass reduction (T. Fitzgerald, S. Sinclair 2022, *pers coms.*). FoMP however, have expressed concerns about weedy grasses on the site and an associated decline in its structure, particularly since 2015 (B. Tomkins 2022, *pers coms.*).

Parks Victoria's management strategy draws heavily from the initial recommendations of Anne Jelinek and Cheryl O'Dwyer (T. FitzGerald 2022, *pers coms.*). Tony FitzGerald, the current site manager (2022, *pers coms.*), is committed to maintaining the current state of the grassland by preventing *Acacia pycnantha* and *Eucalyptus camaldulensis* from encroaching into the grassland and altering its structure. Ecological burning is particularly risky at Mt Piper, as it threatens lichens and fungi that provide important food sources for both threatened *Acrodipsas* species as well as their host ants (DSE 2004) and could cause long-term structural damage to a population of Southern Grasstree (*Xanthorrhoea australis*) (S. Sinclair 2022, *pers coms.*). FitzGerald (*pers coms*, 2023) is interested in exploring opportunities to engage Taungurung cultural burning practices at the site. Accordingly, Mt Piper is carefully protected from wildfires by a series of external firebreaks (DSE 2004). Fitzgerald (2022, *pers coms.*) considers woody vegetation encroachment to be the primary threat to the grassland, and clears woody species manually, according to a five-year cycle, via a combination of slashing seedings and cutting and painting saplings with herbicide. FoMP do not consider these measures to have been sufficient to protect this grassland and have observed high levels of sapling recruitment and accumulating course woody debris (B. Tomkins 2022, *pers coms.*). Anecdotally, FoMP members have observed changes to kangaroo grazing as a result of the encroaching woody vegetation and are concerned that this is causing structural changes to this derived grassland (B. Tomkins 2022, *pers coms.*).

South African Weed Orchid (*Disa bracta*) first invaded the grassland in 2014, and now requires three annual days of hand removal (T. FitzGerald 2022, *pers coms.*). St John's Wort similarly requires control with selective herbicides (T. FitzGerald 2022, *pers coms.*). The site is currently free of *N. neesiana*, making it imperative that future management works do not introduce its seeds to the site (S. Sinclair 2022, *pers coms.*). FoMP are concerned by a lack of government funding to protect Mt Piper (B. Tomkins *pers coms*. 2022). In contrast with New *et al.*'s (2007) initial hopes for this reserve, there has been a steady decline in the resources available to maintain the *S. plana* habitat on the north-eastern face of the mountain, due to a shift in external conservation interest towards the *Acrodipsas* butterflies (B. Tomkins 2022, *pers coms.*). Neither of the *Acrodipsas* species has been formally recorded at Mt Piper since the Threatened Butterfly Community was first listed (K. Boehm 2022, *pers coms.*). Recently, the DELWP Nature Fund have issued a grant to the Threatened Species Conservancy to begin to address some of these concerns. However, Mt Piper still remains in need of broader external conservation funding and industry support (B. Tomkins 2022, *pers coms*). *S. plana* management must be informed by a comprehensive understanding of the broader biodiversity present at a site, and its conservation needs. Such concerns are particularly important within derived grassland sites that do not conform to a standard temperate grassland landscape.

E6. Mt Ridley Conservation Area

Mt Ridley provides habitat for a large and interconnected *S. plana* population that remains largely undocumented and occupies multiple areas managed by different organisations. This area demonstrates the potential for VVP landscapes to support large new *S. plana* populations as well as the difficulties in applying this to management across multiple management zones. Mt Ridley Conservation Area, Craigieburn, lies within Melbourne's northern growth corridor (DSE 2009). The crown-owned area is managed as a series of conservation reserves maintained by Hume City Council (HCC) and Parks Victoria (Backstrom & Forbes 2019; E. Stone 2022, *pers coms.*) among other parties.

Mt Ridley’s VVP Grassy Woodland and VVP Temperate Grassland systems include sites of State Significance (HCC 2013) that support *Dianella amoena*, River Swamp Wallaby Grass (*Amphibromus fluitans*) (Backstrom & Forbes 2019) and Emu Foot (*Cullen tenax*) (HCC 2013). The site retains some of the largest and most intact *Eucalyptus camaldulensis* woodlands in north Melbourne, which provide nesting hollows for a range of native fauna (Backstrom & Forbes 2019). In pre-European history, cultural burning likely played a key role in maintaining the structure of Mt Ridley’s native grassy ecosystems (Abzeco 2007). The former Shire of Bulla’s Inter Urban Break Plan (HCC 2013) afforded Mt Ridley some protection from the intensive urban development and vegetation clearing that occurred across the surrounding landscape, allowing sensitive biodiversity to survive across the site (Backstrom & Forbes 2019). Mt Ridley and its associated landscapes are loosely linked, via Malcolm Creek, to Craigieburn Grassland in the south, though are surrounded by urban and agricultural development to the north, east and west (Abzeco 2007; HCC 2013). Several large blocks of this remnant habitat were assigned to HCC and Parks Victoria in 2001 as a result of the Craigieburn Bypass Development Proposal hearings (*Figure* 13, HCC 2013).

S. plana is abundant across Craigieburn and Mickleham, where it occupies intact areas of grassland and comprises a large and relatively well-connected meta-population (Practical Ecology 2019; E. Stone 2022, *pers coms.*;). Several grassland reserves within this landscape are specifically dedicated to *S. plana* conservation (Backstrom & Forbes 2019). These small reserves have stringent management requirements for *S. plana* (Backstrom & Forbes 2019), in order to meet specific state-level protection targets (DSE 2009). This *S. plana* meta-population is significant due to its isolation from Melbourne’s other S. *plana* populations within larger grassland reserves (DEPI 2013). Most of the *S. plana* habitat present within Mt Ridley is not listed within the Victorian Government’s protection targets (DSE 2009; E. Stone 2022, *pers coms.*). This area predominantly encompasses Mt Ridley Grassland (1. Parks Victoria) (The Author 2020, *pers obs.*), Mt Ridley West Woodland Conservation Reserve (2. Various management) (SMEC 2009, *cited in* Mueck 2012) and Mt Ridley Extension (3. Parks Victoria) (The Author, 2020, *pers obs.*) (*Figure* 12).

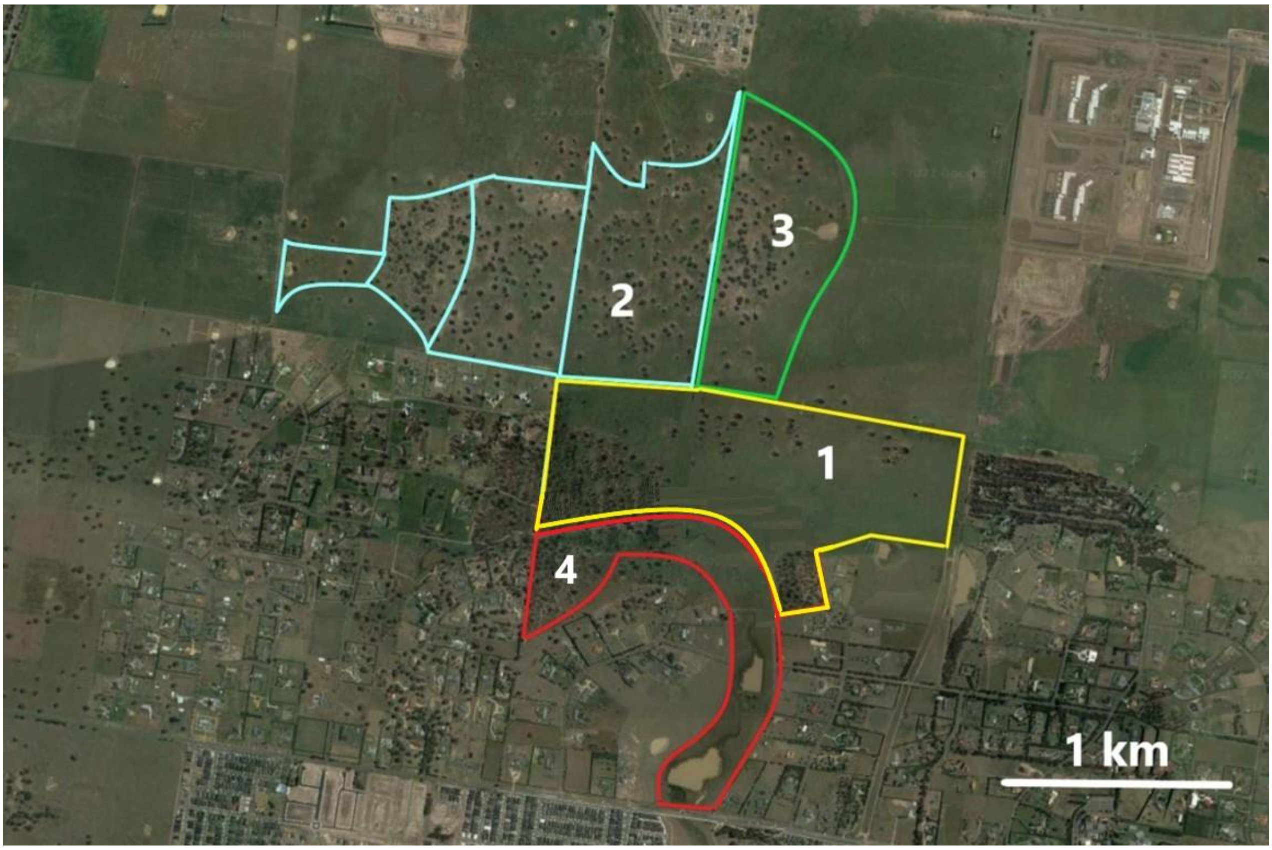

Figure 12. Mt Ridley Conservation Area and its four main management blocks; 1. Mt Ridley Grassland (Parks Victoria), 2. Mt Ridley West Woodland Conservation Reserve (Various, formerly Evolve Development), 3. Mt Ridley Extension (Parks Victoria), 4. Mt Ridley Nature Reserve (HCC).

S. plana was first recorded at Mt Ridley in 2009, within the eastern blocks of Mt Ridley West Woodland Conservation Area, prompting specific management of the woodland towards a suitable site structure for *S. plana*, which began in 2012 (SMEC 2009, *cited in* Mueck 2012; DEPI 2013). A VBA review by the Victorian Government found no prior records of *S. plana* within the adjacent HCC-managed block; "Mt Ridley Conservation Reserve" (4., *Figure* 12) (DEPI 2013). The two Parks Victoria-managed areas were not considered viable for protection as *S. plana* habitat, due to uncertainties regarding their future status (DEPI 2013). HCC listed *S. plana* as a potentially relevant threatened species for their site, anticipating that restoration works would provide general benefits to *S. plana* over the next decade, should the species be present, by improving the general grassland structure of the block (HCC 2013).

In 2019, land management contractors began to observe *S. plana* within Mt Ridley Extension (ECA 2022; The Author, *pers obs.*). A subsequent presence/absence survey recorded 49 individuals during a two-hour survey (ECA 2022; The Author, *pers obs.*). High numbers of *S. plana* were recorded the following year across Mt Ridley's central grassland, including large numbers of males within the HCC reserve flying along slashed firebreaks (The Author 2020, *pers obs.*; ECA 2022). These records were assumed to constitute a single contiguous *S. plana* population, utilising the grassland habitats across Mt Ridley (The Author 2020, *pers obs.*). Eric Stone, HCC's Conservation Team Leader (East) (2022, *pers coms.*), considers specifically targeted *S. plana* management to be unsuitable for the broader biodiversity values present within the reserve, due to the high disturbance regimes required, the restrictive burning requirements and the existence of focused *S. plana* management programmes within nearby sites. The presence of this *S. plana* population nevertheless provides him with a unique opportunity to compare the results of specific *S. plana* conservation approaches elsewhere in Melbourne, with the more standard ecological management that he deploys at Mt Ridley Nature Reserve (E. Stone 2022, *pers coms.*). Eventually, HCC will formally survey their portion of Mt Ridley's *S. plana* population, which will allow them to quantify its site occupancy in detail (E. Stone 2022, *pers coms.*).

Mt Ridley's *S. plana* habitat is threatened by a range of perennial and annual weedy grasses, including Cocksfoot, Yorkshire Fog and Perennial Ryegrass, that damage the habitat structure of these grasslands and out-compete native food plants needed by *S. plana* (Backstrom & Forbes 2019). Key weeds within the HCC Mt Ridley Conservation Reserve, including Gorse, Spear Thistle and Artichoke Thistle, must be manually controlled with herbicide (HCC 2013). *S. plana* is also impacted by soil compaction, caused by horse riding and illegal trail biking (M. Cusack, Parks Victoria, *cited by* Abzeco 2007).

Parks Victoria and HCC are managing their areas for different purposes. Parks Victoria is for conservation, while the HCC managed area is public open space that currently facilitates passive recreational use (walking). There are opportunities for the two agencies to liaise on how to manage this vast landscape. HCC maintain suitable biomass levels at the highest quality areas of this landscape using small-scale ecological burns (Backstrom & Forbes 2019; E. Stone 2022, *pers coms.*). Parks Victoria deploy broadscale burns, on a 3-4 year schedule, on VVP grassland areas of Mt Ridley that they manage (Abzeco 2007; Mueck 2012; E. Stone 2022, *pers coms*) and a slightly longer schedule for woodlands (Abzeco 2007; Mueck 2012). These woodlands require eucalypt thinning, because their natural fire cycle was disturbed during their use as farmland, which has resulted in *E. camaldulensis* recruiting at an unnaturally high density, which threatens the grassy understorey structure (Abzeco 2007; Mueck 2012). Parks Victoria initially planned to deploy sheep grazing as a supplementary biomass reduction measure within their Mt Ridley sites, though this proved logistically untenable at that time (Abzeco 2007; HCC 2013). The future habitat and occupancy of *S. plana* at Mt Ridley will be influenced by the individual management approaches deployed within each block of the habitat, and the extent to which their managers can integrate their approaches across the broader site.