Biomass thresholds for species dominance and diversity in sown urban meadows

Is it possible to determine what happens in the longer term through design?

BY:

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A THESIS SUBMITTED TO THE UNIVERSITY OF SHEFFIELD
IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR
THE DEGREE OF DOCTOR OF PHILOSOPHY

DEPARTMENT OF LANDSCAPE ARCHITECTURE
FACULTY OF SOCIAL SCIENCE
THE UNIVERSITY OF SHEFFIELD

NOVEMBER 2020

Abstract

Naturalistic meadows inspired by the appearance of grassland communities in nature have been increasingly fashionable as a design tool in urban landscapes. Forb species richness and diversity in a meadow community often determine public appreciation in urban areas and the ecological values to pollinating invertebrates. However, grasses are often more competitive leading to dominance in the longer term particularly on fertile soils. Where grasses are excluded in seed mix design to try to slow down this process, weeds tend to be more problematic, potentially leading to parallel declines in forb species richness and standing biomass. This study aims to test the possibility to design meadow mixes that contain grasses but that do not lead to competitive elimination of the forbs at least in a short term. The research involved setting up a field experiment at the Green Estate Ltd, Manor Top, Sheffield, UK. Twenty nine forbs (15 species geographically distributed in both Western Europe and Inner Mongolia, China and 14 species distributed in Inner Mongolia only) and one grass species (*Deschampsia cespitosa*) were established by sowing *in situ* to test design variables: initial sowing rates (including 2 sowing densities and 3 sowing ratios of forb to grass), 2 ecotypes of *Deschampsia cespitopsa* (short and tall to represent a gradient of light competition), and 2 contrasting depths of sand mulch layer (i.e. difference in levels of moisture stress). The study showed that a forb (predominantly *Achillea millefolium* and *Echinops sphaerocephalus* in this study) dominated community can be achieved thought design, but sensible management is essential to maximise performance. Light competition and especially the ability to project taller foliage above grass canopy before it closes the ground proved to be a critical factor. However, dominant forbs reduced the diversity and abundance of subordinate forbs through competition in the same way as grasses do. Thus, this study suggests that species characteristics can play a more important role than the life form of forb or grass in a community.

Increasing forb sowing density or decreasing grass sowing density increased forb seedling numbers but not forb biomass in the longer term. Also, having a higher forb sowing rate speeded up the dominance effect. Thus, the most powerful design tool for a forb rich and diverse meadow community is not sowing density or ratio of forb: grass across all species but changing the ratio between the species, and in particular reducing seeds of the potential dominant species seeds to very low levels. The higher moisture stress increased the numbers of forb seedling and biomass but decreased the subordinate forb biomass. This may benefit forb species diversity in the short term but reduce the biomass coexistence in the longer term. Forbs that had a distribution in both Western Europe and Inner Mongolia appeared to be more persistent than Inner Mongolian forb species, with ability to compete for light more important than geographical origins per se.

An overview to the designed meadow community involved in this research project (picture taken on the 29th July 2019)

Declaration

I, Mingyu Jiang, confirm that the Thesis is my own work. I am aware of the University's Guidance on the Use of Unfair Means (www.sheffield.ac.uk/ssid/unfair-means). This work has not been previously been presented for an award at this, or any other, university.

Acknowledgement

Throughout this study I have received a great deal of support and assistance from a number of wonderful people to make it possible.

First of all, I would like to thank my supervisor Professor James Hitchmough for his invaluable supervision and comprehensive support in all aspect. I have been extraordinarily lucky and honoured to be working with a such encouraging and exceptionally inspiring supervisor who made my PhD journey an incredible experience and even positively influenced me far beyond this research project. Besides, thank you for spending so much time on my field work and thank you for always saying ‘it’s not the end of the world’ when I feel things get difficult.

I would also like to express my thanks to all staff in the Department of Landscape Architecture, especially Ms. Helen Morris for her patience and helps. Also, big thanks to Dr. Jean Russell at the University Research Support Team for her advices on my statistical analysis. Special thanks to everyone I met at the Green Estate ltd particularly Ms. Sue France for sorting out everything on my experiment field.

Many thanks to my PhD colleagues on the 9th Floor, the Arts Tower. Without their voluntary help my trial plots would not be physically set up. Also I wish to say thank you to those of my ‘lunch mates’ for sharing their thoughts and encouragement during the mealtime.

Last but not least, I must acknowledge all undoubted support from my mother Mrs. Hongyu Qu and my father Mr. Wenchao Jiang. I really appreciate they encourage me to choose and respect the path I take, as well as to my wife Mrs. Echo Xiaoyu Zhu for her company and sharing.

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Chapter 1 Introduction

1.1 Background

Semi-natural grasslands, inspired by the complexity of more natural herbaceous communities, have become increasingly fashionable in both Western and Eastern countries as a design tool for landscape to improve both social and ecological values (Hitchmough and Dunnett, 2004; Jiang and Yuan, 2017). In urban settings, the public's social acceptance is of parallel importance to ecological persistence of species for sustaining a naturalistic herbaceous community, especially where these landscape types are not familiar to most lay people compared to conventional mown grasslands (Southon *et al.*, 2017). Community performance in line with initial design expectations are essential to deliver the cues such as care and excitement that are expected by the public and which help gain public support (Nassauer, 1995; Hoyle, Hitchmough and Jorgensen, 2017a). The urban public appears to becoming increasingly biocentric, however, the ecological benefits of meadows may fail to be perceived without clear design intentions (Southon *et al.*, 2018). A high ratio of forbs to grass leading to high density of flowers have been recognised as critical indicators to deliver these desired notions. Highly flowery meadow communities have been shown to be one of the most appreciated greenspace type to the public (Southon *et al.*, 2017; Hoyle *et al.*, 2018).

In a grassland ecosystem, maximum biodiversity values are usually achieved where forbs diversity and biomass are high (Walker *et al.*, 2004; Del-Val and Crawley, 2005; Pywell *et al.*, 2007). However, particularly in early industrialised countries, most current day grassland habitats often have a history of agricultural fertilisation which increase grass species dominance (Pywell *et al.*, 2002, 2007; Sluis, 2002; Walker *et al.*, 2004). Grass species are consistently the superiors in the system due to abundant seed recruitment from soil seed banks, and vigorously vegetative growth and persistence etc. (Pywell *et al.*, 2003). Forb rich grasslands mostly exist in the part of the world where has been historically managed by traditional methods (e.g. grazing or hay harvest) (Hitchmough, unpublished). However, the forb rich communities are fragile in that species diversity soon disappears when the management ceases. The ecological process that support forb richness is unlikely to be recovered simply through reintroducing the original species (Sluis, 2002). Restoration is theoretically feasible but is potentially an extremely slow process under productive soil conditions (Walker *et al.*, 2004). Re-establishing the equilibrium between forb and grass biomass may require continuous management approaches for few decades (Silvertown *et al.*, 2006). The general experience in ecological practice is that even when forbs are initially dominant, this situation generally shifts to dominance by grasses within a few years (Del-Val and Crawley, 2005; Dickson and Busby, 2009).

Planting design in urban landscapes can provide a clearer starting point (i.e. sowing into a ground in which the weed seed bank has been supressed and closely managing competition within the community from the beginning. However, grass dominance that reduces forb biomass and diversity is still a common problem. Where grasses are left out from the species mix to slow down the process of grass dominance, weeds tend to be problematic in the meadow also leading to a decline in forb performance (Hitchmough, Paraskevopoulou and Dunnett, 2008; Hitchmough, 2017a). This is also the major concern in projects proposed and managed by the Green Estate Ltd, one of the experimental sites sponsor in this study.

Bjørn, Weiner and Ørgaard (2016) questioned whether it was possible to create a sustained forb rich community through design. Dominance and invasion seemed to inevitably diminish the original design effects unless intensive management is involved. From the other prospective, designing the outset to maximise the initial ratio of desire species to weeds appears able to enhance the community persistence in the long term. These approaches usually involved topsoil removal and application of unproductive mulch layer (e.g. sands) (Hitchmough and Fleur, 2006). Also, using tall productive species that have broadly equivalent competitiveness to weedy grasses can resist the community from invasion in the long term (Hitchmough and Fleur, 2006). However, designs contain a base level of grasses can form a valuable contrast to brightly coloured forbs and create a feeling of coherence and relaxed naturalness (Lindemann-Matthies and Bose, 2007; Lucas, 2011). The grass biomass can also close the gaps in an early stage before the forbs are established when the selected forbs are significantly less productive. This raises that question that whether it is possible to design meadow mixes that contain some grasses but that do not at least in the short-term lead to competitive elimination of the forbs? Also, is there a critical grass biomass threshold for decline in forb biomass and diversity?

Colour diversity is a critical indicator for ecological benefits to the public (Hoyle *et al.*, 2018), a forb dominant community also requires a good level of forb species coexistence that contains sufficient biomass of many flowering species rather than merely forb survivals which are visually insignificant. However, species loss and the trend of asymmetric development through both inter and intra species competition are naturally part of the ecological process (Grime, 2002). The pattern of dominance by few species is often long retained (Silvertown *et al.*, 2006). Practical experiences also suggest the factor of ‘species’ may be more impactful and even override other design factors such as plant density (Hitchmough, Wagner and Ahmad, 2017; Bjørn *et al.*, 2019). Thus, we do not expect a highly symmetric growth within the communities, however, a relatively high coexistence and establishment of diverse species can be achieved through design. Even where diversity may only be retained in a short term, it is still socially valuable (Hitchmough, Paraskevopoulou and Dunnett, 2008) and potentially allows scope for further management to be applied to extend the community values.

Thus, for landscape practitioners, the core technical question is can what we design determine what is established at least for a short term (e.g. three years)? In herbaceous communities created by sowing techniques, longer term species composition is unlikely to reflect the community composition beyond this, with competition within the community and invasion from outside take place, community appearance can quickly become something very different from the original starting point. Most commonly, any subtle difference in sowing density and soil productivity may significantly affect the competitive relationship and drive the community to a completely different direction. This research will mainly focus on how the starting point of those two factors affect the forb diversity within community and how grass biomass shapes this in the longer term.

Most of the ecological literature on forb establishment and grass competition involves native forb species (for example, Pywell *et al.*, 2002; Walker *et al.*, 2004; Del-Val and Crawley, 2005). Much less is known about how non-native forb species compete with grasses, and whether they are more or less sensitive than native species or show no such patterns. Non-native species have been proved to have equivalent ecological values as native species for at least some invertebrates (Owen, 2010) and the capacity to improve landscape values (Hitchmough, 2011, 2017a). This study has an implicit cross cultural basis in that it deals with meadow vegetation both in the UK and in Inner Mongolia, with the added complexity that many Inner Mongolian grasslands species are also UK native species. This raises some interesting research question; are some Inner Mongolian forbs with a European distribution equally fit to grow within the UK climate in competition with grasses, and are Mongolian forbs with purely Eastern Asian distribution less well fitted in the UK? Or, are canopy heights and layering far more important in determining competitiveness and the outcomes?

This study involves setting up controlled field experiments in order to test how these factors interact in designed meadow vegetation in order to assess how theoretical principles on grass competition might be translated to create a practical model for future meadow design.

1.2 Research questions

Key overall research questions for this study are as follows;

- Do grasses inevitably dominate sown meadows leading to a decline in forb survival and biomass?

    - What are the impacts of grass competition on forb biomass and survival in sown meadow-like grasslands?

- To what extent, can design approaches enhance forb performance in relation to grass competition?

    - Does increasing sowing density lead to grass dominance happening sooner?
    - Does initial sowing ratio of forb: grass affect forb survival and abundance of forb biomass in the longer term?
    - Does grass density and foliage height affect forb suppression?
    - Can grass dominance be reduced by increasing nutrient and water stress?

- Can a persistent forb rich and diverse community be achieved (with reduced effects of dominance) in urban landscape through design?

    - To what extent, can design approaches enhance coexistence within meadow forb communities?

- What forb species tend to be more persistent?

    - Are non-native forbs (i.e. forbs not native to the UK) more subject to competitive dominance than native forbs?
    - Does increasing the height of the forb canopy reduce the capacity for grasses to dominate in meadows?

- Is there a critical grass biomass threshold for decline in forb biomass and diversity?

Chapter 2 Literature Review

2.1 Perceptions to grasslands across a contextual gradient from natural to urban environments

2.1.1 Perception of grasslands at the landscape scale

Grasslands are the largest geographical vegetation type found on every continent (with exception of Antarctica) covering about 40% of land surface (Allaby, 2006). Ecologists define as perennial grass dominated communities as those with few or no scattered woody species, and these are an important part of the ecosystem (Spedding, 1976; Allaby, 2006).

Grasslands played a central role in human evolution. East Africa gradually shifted from dense forests into savannahs from about 14 million years ago (Allaby, 2006). Krajick and Lee (2016) argued that this evolution of grassland, creating a large open landscape, provided a more diverse sets of niches and opportunities for hominids to develop their living and social skills, for example to co-operate with one another in hunting and foraging. Hominids evolved to become human-like in East Africa from 6 million to 7 million years ago. Grassland became geographically dominant due to its flexibility to adapt to environmental disturbance - as did humans. After hundreds of thousands of years of living as hunter gatherers, a profound revolution to human's way of live began from around 13,000 years with the cultivation of crops and domestication of animals (Diamond, 2002). This transformation settled, civilised and expanded human population (Diamond, 2002). Settlements gradually expanded and became permanent, and humans became more alienated from the grasslands they had originally evolved in (Morris, 1994).

The physical alienation to the grasslands shifted the 'perceptible realm' (Gobster *et al.*, 2007) and unfamiliarised the original habitat of human beings which raised the notion that grasslands are the 'wild' (Evanoff, 2005). Humans also culturally shifted, particularly post the industrial revolution from being a 'grassland species' to a 'urban species' and lost opportunity to maintain close proximity to wild nature (Turner, Nakamura and Dinetti, 2006). In the early 19th century, American prairies were still referred as 'internal deserts' due to being 'unfit for cultivation' (Kinsey, Roberts and Sayre, 2011). These attitudes and the need to generate food to fuel the industrialisation of Eastern America, led to the destruction of the native people grassland cultures of the USA in the C19th.

In other parts of the world, cultures that had experience much longer periods of contact with other cultures continued to operate as grassland dependent people. In Northern China, Traditional Mongolian culture has remained dependent on grasslands up to the present day, and the same is true for many other cultures in Asia, Africa and South America. These cultures involve a more interactive and an all-encompassing system between humans and nature. Using animals and plants as living resources through herding and transhumance are seen as moral activities that also contribute positively to maintaining grassland ecosystems. Grasslands are perceived as the homeland for Mongolian people because they offer freedom over life styles and freedom of mobility (Humphrey, Mongush and Telengid, 1993).

From grassland aesthetics point of view, Mongolian grassland people value the openness of grassland landscapes and accept the various range of plant communities as it is ‘what it is’ (Humphrey, Mongush and Telengid, 1993). The public living away from grasslands perceive grasslands as ‘prospect’ and ‘picturesque’ (Kinsey, Roberts and Sayre, 2011). They appreciate the ‘open glades with smooth ground texture’ as the image of grassland in people’s mind (Ulrich, 1986). Grasslands are of course a mixture of grass and forbs, although typically dominated by the former. Forbs potentially add more colours and dynamic change to grasslands. However, at landscape scale, people are unlikely to notice the difference at species level and forb richness is likely to be visually less important to humans living in these landscapes.

2.1.2 Perception of rural grasslands

Today, grasslands in the rural areas of post-industrial societies, other than those in nature reserves are almost all managed for various agricultural purposes. In the UK, although they mostly exist to support agricultural activities from the very intense, for example silage production, through to low intensity grazing and hay making, typically in the uplands under more marginal conditions. This diversity or gradient of roles means it is widely accepted that they have social and ecological as well as agricultural values (Davies *et al.*, 2006). In China, there has recently been policy moves to in a similar direction aimed at restoring original grassland values (Wang *et al.*, 2018).

In many countries, most rural grasslands we see today are heavily transformed from their natural antecedents through agricultural management. Grasslands are easier and cheaper than other types of land (e.g. woodlands) to transform for new agricultural uses, such as to grow new crops. European countries converted woodlands, and heathlands to grasslands for agricultural uses centuries ago (in some cases millennia ago) while the US began farming tallgrass prairie only in the 19th Century (Allaby, 2006). Today, rural grasslands for the most of Europe are semi-natural and represents cultural landscape (Gobster *et al.*, 2007). In China, increasing livestock rate on semi-natural grazed grasslands was designated as a key political principle from the 1950s on (Cao *et al.*, 2013). The impact of this has been to reduce species diversity, and in particular diversity of forb species in these grasslands. Structural and species changes in these grazed grasslands have been gradual and whilst these changes have resulted in grasslands that are very different from those which existed at least a century ago, the public may have already adapted to and accepted these. It is just like urban people preferring close mown grass because they are familiar with the tidiness (Hoyle *et al.*, 2017). However, this familiarity may consist of a cultural loss of rural experience and aesthetic richness.

Intensively farmed grasslands are found all over the world especially in developed countries where more advanced techniques and greater demand of agriculture productions are in operation. After the industrial revolution, improved transportation and agricultural technologies provided people more possibilities to utilise grasslands that were once considered distant and 'unfit for cultivation' (Kinsey, Roberts and Sayre, 2011). Also ownership of specific areas of grassland, especially in the US, enhanced the sense of belonging (Kinsey, Roberts and Sayre, 2011). These changes were likely to make grasslands more physically accessible and psychologically close to humans again. Agriculture made rural grasslands more familiar, controllable and safer to human. This awareness of safety and coherence reflected human needs of grassland (Gobster, 1995). For the grasslands that were most distant, people became more enthusiastic towards the 'internal deserts' (Kinsey, Roberts and Sayre, 2011).

Aesthetic preference to agricultural grasslands to rural people

People's demands on grasslands as resources influenced their aesthetic preference for these agri-ecosystems. Agricultural people living in rural area may appreciate grasslands as crop lands or swards implying the sense of 'productivity' and 'utility' (Williams and Cary, 2001). Well manged or 'neat' crop lands may stand for a sign of prolificacy, even wealth. Neatness, represented by the landscape picture of tall, uniformly green and weed-free, is a strong predictor of the attractiveness of rural agriculture landscapes (Nassauer, 1992). For feeding livestock, intensive agriculture tends to value grasses more than forbs although many forb species are palatable for livestock, and more protein rich (McIntyre, McIvor and Heard, 2004). As a result of this grass only culture, forbs are usually not desired or even treated as weeds in rural grasslands by farmers (Cook, 1983). As result of these attitudes and the removal of forb biomass by grazing has led to severe degradation of diversity, and in particular forb diversity of grasslands.

Needs for delivering ecological and aesthetical notions to the urban public

Parallel to the development of attitudes to grassland as simply an agricultural 'utility', concern of loss of natural beauty, which implies ecological value in rural landscapes, became popular among the public in western countries from the 1960s. This awareness was probably due to increasing urbanisation in many developed countries and a series of environmental crises and perceptions of crises in the US (Gobster, 2010). This led to the notion that grasslands should be both abundant in productivity and offer ecological benefits as rural grasslands (Lee-Hsueh, 2018).

However, it is reasonable to assume that the way that public perceive the signs of naturalness or ecological benefits are different from ecologists. Williams and Cary (2001) found that there are ‘considerable challenges’ to communicate the importance of less productive grassland ecosystems preferred for nature conservation with the general public. Ecological services and aesthetic attractiveness of these systems may not be judged to be aesthetically pleasing to the lay public (Lee-Hsueh, 2018). The height and complexity of vegetation structure on grasslands are important factors for the sense of naturalness to the public. For example, areas with dense trees are seen as more natural than open grasslands; tall and messy grasslands are seen as more natural and ecological (Williams and Cary, 2001). Yet less productive grasslands, despite of high species diversity, may be less valued by the public. Species diversity is an important index in ecology, but people tend to underestimate it regarding the herbaceous performance in a grassland system (Lindemann-Matthies, Junge and Matthies, 2010).

In rural context, the sense of either productivity or ecological values is widely accepted in different locations in rural grasslands. Urban people who visit rural grasslands mostly expect to be away from the routine environments to natural settings for restoration while restorative effects are predominantly perceived from a large scale of rural landscape rather than a micro scale of species composition (Kaplan and Kaplan, 1989; Kaplan, 2001). Also, the urban public is unlikely to expect the notion of cultural realm in rural landscapes. Remote landscapes are usually perceived as somewhere beyond the territory that requires the notion of duty of care. Vegetation in rural grasslands are aesthetically expected to reflect rural characters with less needs for excitement and the sense of familiarity (Nassauer, 2011).

Rural grasslands in the UK

In the UK, agriculture is extremely intensive and dominates rural landscapes in the 20th Century. Agricultural land covers approximately 70% of the total UK land area (Downing and Coe, 2018) and politically has been encouraged by the Common Agriculture Policy (CAP) from the EU. The CAP subsidies can make up about 50% - 80% of UK farmers’ income. This support is unlikely to be affected after the Brexit and will still be offered at least until after 2024. The contribution of the agricultural industry in the UK is significant and provides for a certain degree of food security, export values and jobs for about 466,000 people (Downing and Coe, 2018).

The application of fertilizers pre and part of the CAP to boost agricultural production caused massive loss of species diversity in UK grasslands. It began to decrease in the 19th Century when the fertiliser was applied to increase agricultural productivity. This became severe during the 1940s and 1950s due to uses of chemical fertilisers and herbicides (The Wildlife Trust, no date). Additional fertiliser increases soil productivity which boosts for competitiveness of the most productive species many of which are grasses, which then suppress the performance of many stress-tolerating forb species (Grime, 1988). The greater the fertilizer use the greater the biodiversity loss.

In the C19th meadows could contain up to 40 species per square metre but few habitats can match this diversity in the UK today, due to excess fertiliser usage (Ridding, Redhead and Pywell, 2015). During the 20th Century, over 97% of species rich wildflower meadows have been lost, while only 1% of total UK land area today is species rich grassland (Coles, 2015). This dramatic change to the UK rural landscapes began to happen more than a hundred years ago and undoubtedly transformed the public impressions of what is normal in grassland landscapes. Familiarity with landscapes determines the preference of landscape scenes (Orland, 2013); hence publics who have never seen landscapes of species rich grasslands readily adapt to the current reality of mostly low diversity grasslands. Therefore, it is reasonable to assume that, currently, species poor rural agricultural landscapes have been widely accepted by the public as normal. The disappearance of grassland diversity is part of an unseen loss of cultural experience in British rural landscapes. Increasingly there is growing awareness of this situation, and public interest to beginning to revive, for example, through events like The National Meadow Day in every July (Save Our Magnificent Meadows, 2019).

Grasslands in China

The grasslands of Inner Mongolian, China are the largest existing contiguous grassland biome in the world. It has also suffered from degradation in terms of species diversity from the 1950s; 90% of grasslands have degraded in both the size of grasslands and the volume of biomass in the 2000s (Liu *et al.*, 2018). The same pattern of loss happened in other grassland communities in China during this period. Development of a more intensive agriculture grassland industry in the UK was driven by a combination of the individual choices of farmers responding to agricultural policy within a market-oriented economy. In China, the current situation in Inner Mongolian grasslands is largely a response to highly centralised political decision making (Conte, 2018).

The idea of grassland as ‘waste’ was derived from a Maoist vision from the 1950s (Marks and Shapiro, 2002). A series of political campaigns which aimed to maximise the utilisation of grasslands, mainly about massively cultivating grains and vegetation removal for construction of infrastructures, caused the most grassland degradation (Cao *et al.*, 2013). The stocking rates in the remaining grazing area were also increased with a view to maximise ‘effectiveness’ livestock production. However, the intensive herding still respected traditional nomadism and was not considered as a major threat to grassland degradation until the 1980s (Cao *et al.*, 2013).

In the 1980s, the policies from the central government significantly changed the system of grassland management. The livestock and pastures owned by communes were decollectivized and divided among pastoral households. This policy was generated by recognition of the ‘tragedy of the commons’ model. However, in Inner Mongolia, the policy of enclosed pastures and limited livestock’s mobility significantly lead to overgrazing and intensive trampling (Cao *et al.*, 2013; Conte, 2018). What commonly happened was that necessary (to maintain diversity) appropriate ecological disturbance was absent in productive areas while overgrazing took place in unproductive areas. This massively changed the original ecological pattern which were based on organically derived understanding from practice in combination with Mongolian philosophy and traditionally derived knowledge of nature.

Today, it is unclear whether people living in these areas mourn the loss of species diversity grasslands. As in the West it seems that people living in grasslands have adapted the current livestock production system and accept the land use under the combination of the current political framework and market forces (Conte, 2018). Again, human familiarity and adaptability facilitates the acceptance of new, degraded landscapes as normal.

However, the cultural loss in historical grassland ecosystems in China is probably more serious compared to western countries. It seems reasonable to assume that forbs and their flowers were more symbolic and appreciated in China, although there is no research on the comparison of preferences for flowers in rural landscapes within western and Chinese culture. The images and symbols of flowers usually have more comprehensive and deeper cultural meanings in historically pastoral regions of China while they stand more for horticulture, gardening or appreciation of nature in post-industrial western cultures. A good example is that, in Chinese literature, flowers regarding their habitats and traits are commonly personified to represent the characters of people (e.g. Lotus stands for the personality of integrity regarding the Chinese phrase of ‘growing out of mud, the lotus blooms pure and untainted’). Due to these positive associations, images of flowers are very popular in traditional paintings, to appear on book covers or to be used as patterns on textiles as part of spiritual visions.

Fortunately, from the mid-2000s, China central government recognised these problems and launched a series of programme, known as ‘Beautiful Countryside Construction in China’ (Liu, 2015). An important part of this programme is to restore the rural landscapes with local characteristics and using native wild species to reveal the landscape characters. This is a good opportunity to bring the ‘past’, when grassland communities in many rural areas were species-rich, back to the ‘reality’. In the meantime, ‘Restore Pasture and Retire Livestock’ (reducing agriculture intensity) policy was applied in grassland regions including Inner Mongolia (Cao *et al.*, 2013). However, simply excluding the grazing, which is again contrary to the nomadic philology on ecology, appears less effective on restoring species diversity and biomass on grasslands (Wang *et al.*, 2018). Although the impact of this policy has to some extent been criticised as ‘overstated’ (Yeh, 2010), these are still good political foundations to enhance the species diversity in the context of rural grassland in China. What remains blurred is that there is no clear guidance on restoring the herbaceous community of a grassland system e.g. cultivation approaches and management vision for the longer term.

2.1.3 Grasslands in urban areas

Nowadays in the UK, grassland normally covers up to a third area of a town or city while about two-thirds is managed as mown grasslands primarily for recreational and social purposes (Forest Research, 2020). Mown grassland as urban green infrastructure is also accepted by and attached to the aesthetics of the urban public due to the familiarity of the landscape type (Nassauer, 2011). Frequent mowing can maintain the tidiness and sense of care to the public (Nassauer, 2011). However, ecological are low in mown grasslands due to lack of species diversity and canopy layers to support invertebrates (Smith, Chapman and Eggleton, 2006). This lowers the greenspace quality in terms of ecological services which is recognised as a key aspect in urban landscapes in the 21st Century (Hitchmough, 2004; Marzluff and Rodewald, 2008). Also, whilst neat mown grasslands have a possible restorative effect on people, a lack of colours and signs of biodiversity reduces aesthetical pleasure (Hoyle, Hitchmough and Jorgensen, 2017a). Increasing amounts of research has focused on connections between aesthetics and ecology in urban landscapes in the 21st century, and concludes that landscape design should enhance both aspects for a win-win strategy (Dunnett and Hitchmough, 2004; Gobster *et al.*, 2007; Özgüner, Kendle and Bisgrove, 2007).

Urban meadows, that are unmown and seen as natural or less disturbed grasslands, improve the complexity of community structure. This can be an alternative to mown grasslands to increase the biodiversity values and decrease the management costs. The public's perception of grasslands is far more sensitive for urban than rural grasslands. 'Messy' and 'disordered' grasslands can deliver the signs of ecological benefits in rural settings (Williams and Cary, 2001) but are challenging to be accepted as a part of daily landscape in urban areas (Hoyle *et al.*, 2017; Hoyle, Hitchmough and Jorgensen, 2017a). 'High human intent' is the dominant perception in urban landscapes (Gobster *et al.*, 2007). Simply introducing nature and biodiversity back to the urban landscapes through ecological approaches blurring the boundary between the context of the 'wild' and 'urban' realms can be risky for social acceptance. Designing for both social and ecological needs in urban areas therefore requires understanding of the cultural and philosophical development of relationship between humans and nature. The alienation from nature and interests in naturalistic vegetation took different paths in the Western and Oriental cultures.

Alienation from nature and interest in naturalistic landscapes in Europe and in China

The idea of cultural alienation from nature in the western world was generated within ancient Greek philosophy. People in this time were highly influenced by the idea that only humans were rational beings and only they possessed the capacity for moral choice, which placed themselves in opposition to nature (Jenni, 2005). These ideas were influential until the publication of Darwin's *Theory of Evolution* (1859) (and afterwards). Despite increased alienation between humans and the natural world resulting from the industrial revolution in the 19th Century and dramatic growth of urban areas in the 20th Century (Berry, 2008), practical ideas to revive human appreciation of, and connections with the natural world have been widespread in the western world. *Flower of the Field* (1851) by Reverend C. A. Johns had firstly drawn attention to encourage gardeners to reappreciate wildflowers. William Robinson then build on Johns' ideas and is often considered as an early applier of ecological ideas, especially in connection with his publication *The Wild Garden* (1870). However, this only dealt with the naturalisation of hardy exotic plants in grassy swards and was merely supposed to modernise an old gardening practice (Woudstra, 2004). From the early 20th Century, the emergence and popularity of amateur gardening in Western Europe appears to have contributed to the idea of natural landscapes; spontaneous plant communities were introduced as a model for design also became evident in the Netherlands and Germany (Hitchmough, 2017a). The value of natural environment began to be promoted. From 1970s, landscape designers were influence by this and attempted to reflect the idea in practice (Özgüner, Kendle and Bisgrove, 2007; Kendle and Forbes, 2013). The application of environmental ethic ideas to nature in the urban context to designed communities is therefore a relatively recent phenomenon which has developed spontaneously in many places particularly in the western world during the past 40 years (Goode, 1998; Woudstra, 2004).

Integration of people with nature was a key idea within Ancient Chinese philosophy. From at least the 4th Century BC, Daoism started to develop its environmental philosophy and the aspiration of people to live in a manner convergent with nature (Cooper, 2014; Liu, 2016). However, in most of the following dynasties, Confucianism dominates the mainstream of philosophy in China. Although Confucius emphasised more ethical issues between human than natural environments, overall, there was no sharp philosophical distinction between the human and the natural world (Ivanhoe, 1998; Cooper, 2014). Gardening, presenting Chinese imagining of nature, had long history (from about 1,000 BC) and as a cultural symbol. Chinese gardening culture heavily influenced gardening style in the Western world from the 19th Century on (Fan, 2003; Liu, 2012; Peter, 2013). Cultural interest in of nature-like planting declined with social changes and campaigns starting from the 1900s on due to the chaos associated with the collapse of the Imperial State and then again in the 1970s due to the Cultural Revolution when much traditional Chinese culture was damaged (Xie, 2015; Schopohl, 2016; Stanzel, 2016). With the growth of the bourgeoisie in China from 2000s on (Barton, Chen and Jin, 2013), a new positive engagement with nature has gradually occurred again (Xie, 2015). This agrees with the finding in Hoyle *et al.*, (2017) that appreciation of 'wild' landscapes tends to be higher among the people with higher income who pursue a higher hierarchy of needs and a more sense of nature reflected in landscape style.

Currently, in the UK, only people who regularly access wild environments tend to accept more natural landscapes in their daily life while the frequent users of mown grasslands are more likely to prefer the 'tidiness' (Southon *et al.*, 2017). Compared with the results from recent studies between the UK (Hoyle, Hitchmough and Jorgensen, 2017a; Southon *et al.*, 2017) and China contexts (Jiang and Yuan, 2017), naturalistic meadows appear less appreciated in urban landscapes in China (the research was conducted in Beijing and mostly represents the mainstream of Chinese urban culture; the result could be different if it was conducted in a remotely nomadic culture region for example Inner Mongolia). However, Jiang and Yuan (2017) found the similar pattern that the naturalistic meadows are more valued by the people who have better botanical or horticultural knowledge. Limited gardening opportunities due to the dominance of high rise apartment living, and limited engagement to greenspace management work as a community member may distance people from nature in urban area in China.

Public perception of naturalistic meadows in urban landscapes

Naturalistic meadow vegetation inspired by the appearance of natural grassland communities has great potential to enhance both social and ecological values in urban landscapes (Hitchmough and Dunnett, 2004; Özgüner, Kendle and Bisgrove, 2007). Despite meadow-like vegetation increasingly being developed in urban areas, this still only takes up a tiny part of urban greenspace (Hitchmough and Fleur, 2006) and is unfamiliar to the most of the urban public. Also, people who support to enhance urban ecological values are unlikely to enjoy the natural landscape in urban areas (Garbuzov, Fensome and Ratnieks, 2015). To investigate the public acceptance of this type of vegetation in urban landscapes, research was undertaken by Hoyle (Hoyle *et al.*, 2017, 2018; Hoyle, Hitchmough and Jorgensen, 2017a, 2017b) and Southon (Southon *et al.*, 2017, 2018) mainly based on the Urban BESS (Biodiversity and Ecosystem Service Sustainability) in Bedford and Luton, UK and other sites across the UK representing landscape planting contexts in urban areas. Integrating with previous theories of aesthetics and environmental philosophy in urban landscapes (e.g. Gobster, 1995; Nassauer, 1995; Lindemann-Matthies and Bose, 2007), public perception of naturalistic meadows in urban landscapes was explored. Flowers in meadows are recognised as the key to gain social support.

Forb performance determines social acceptance of naturalistic meadow vegetation

To be seen as appropriate in the urban environments, a sense of intention and care is essential to be reflected in the ‘unstructured’ planting (Nassauer, 1995). Tall grassy meadows are unlikely to provide the sense of excitement and enjoyment even to people who in principle support the ecological landscapes (Garbuzov, Fensome and Ratnieks, 2015). Besides, despite the evidence that urban people have increasingly become more biocentric and realise the importance of ecological values (for example the awareness of species rich vegetation supporting more invertebrates, particularly urban pollinators) **(**Hoyle, Hitchmough and Jorgensen, 2017a; Hoyle, Hitchmough and Jorgensen, 2017b; Jiang and Yuan, 2017), the lay public are unlikely to perceive and enjoy the ecological values from the natural form of vegetation without notable cues (Hoyle, Hitchmough and Jorgensen, 2017a). Also, species diversity is often underestimated by the public without legible visual signs, and even landscape professionals were unlikely able to identify biodiversity at the species level (Hoyle *et al.*, 2018). Social sustainability may largely depend on the clarity of signs of ecological value to public.

The communication to deliver aesthetical and ecological cues are critical. Flowers traditionally induce powerful positive emotions (Haviland-Jones *et al.*, 2005) and in urban landscapes, extremely flowery vegetation reflect the sense of design intentions (Hoyle *et al.*, 2017) delivering the cues of care (Nassauer, 1995; Hoyle, Hitchmough and Jorgensen, 2017a). 'Sense of care' can positively connect people with the surrounding environment and potentially enhance the familiarity with surrounding landscapes (Nassauer, 2011). Flowery meadows can gain positive social support by delivering the key messages in multi-dimensions (Hoyle *et al.*, 2018). Forbs in grassland communities therefore reconnect people with nature in urban areas. Forb richness and performance determines to what degree this takes place.

Naturalistic meadows can be more attractive and exciting than the more traditional, horticulturally managed vegetation to the public when meadow communities are highly flowery (Hoyle, Hitchmough and Jorgensen, 2017a; Southon *et al.*, 2017) whereas increase in grass decreases appreciation (Hoyle *et al.*, 2017). The most colourful meadows are perceived as the most interesting grassland and the most beneficial to insects at the same time (Hoyle, Hitchmough and Jorgensen, 2017b). Southon *et al.* (2017) and Hoyle *et al.* (2018) found that the public tend to use colour as a cue for the evidence of plant species diversity. The level of diversity in colour becomes a measure of the perceived level of biodiversity and ecological benefits (Hoyle, Hitchmough and Jorgensen, 2017a). Besides, forbs can enrich the community structure which also delivers the sense of species diversity and increase the sense of satisfaction to the public (Southon *et al.*, 2018). Meadows that contain structural diversity (i.e. foliage heights) were seen a more preferred grassland type and to provide better ecological benefits than short meadows even when the latter is also colourful (Southon *et al.*, 2017). Based on this evidence, a positive relationship between ecological and aesthetic values can be achieved through designing urban meadows (Lindemann-Matthies, Junge and Matthies, 2010).

Flowery naturalistic grasslands have great potential to improve the site quality and appreciation, and gain more supports from a wide range of people (Southon *et al.*, 2017). The flowery meadow in the BESS project has positively changed people's perception to the public preferring the 'flowery nature' over highly managed vegetation (Hoyle *et al.*, 2017; Southon *et al.*, 2017). However, a flower coverage of 27% or greater is suggested as a threshold for high appreciation (Hoyle, Hitchmough and Jorgensen, 2017a). This requires an abundant even dominant forb biomass to persist in a community in the longer term (Bjørn, Weiner and Ørgaard, 2016). Moreover, the diversity of flowering species is also important. Therefore acceptable coexistence between forb species for the longer term is needed in design and management processes. Post-flowering meadows still imply the sense of 'messiness' (Hoyle et al., 2017). Establishment of forbs species and extending the flowery period are therefore critical to successful utilisation of ecological processes.

Introducing non-native species can prolong the flowering period from early spring to late autumn and enhance community floweriness and resilience, by for example using tall North American forbs (Hitchmough and Fleur, 2006; Hitchmough, Wagner and Ahmad, 2017). From aesthetics point of view, people prefer to value the appearance and variety of flowers in a meadow community over the sense of nativeness. Non-native species with additional flower colours and forms in urban landscapes are potentially more attractive and interesting than native species per se (Hoyle, Hitchmough and Jorgensen, 2017a). The public are also likely to prefer the notion of 'ecology' over a more puritan 'nature' in urban landscapes. They tend to appreciate exotic species when they are well informed with ecological knowledge such as benefits of non-native species to climate change (Hoyle, Hitchmough and Jorgensen, 2017b). Finally, as Britain has long history of importing and cultivating non-native plants (Dehnen-Schmutz *et al.*, 2007), the public are generally culturally accepting of these (Hoyle, Hitchmough and Jorgensen, 2017b). The perception to non-native species is different from rural landscapes and this provides a good opportunity to enrich the species diversity in urban ecosystem.

To conclude, forbs are the key to increase the appreciation to grassland community in urban landscapes. Diversity in colour and structure provides visual pleasure as well as ecological values. Frequently, visible species diversity represented by the flowering species deliver the signs of biodiversity which create the enjoyment and satisfaction to the public since biocentric values are increasingly fashionable and recognised in urban landscapes. However, forb richness and persistence are a challenge for both social and ecological sustainability. A core question would be that can design, that through applying the ecological process and is involved at the beginning, determine the forb performance in both short and long terms?

Table 2.1 The perception of sense of care and intention increases with the gradient from natural to urban contexts. Thus, design becomes an extremely critical issue with the increased sense of cultural realm. Forb richness is an essential indicator in urban landscape designs.

	Grassland at Landscape scale	**Rural grassland**	**Urban grassland**
Perception	Nature of open glades with scattered or no trees	Rural people – agricultural land and utility; Urban people – natural environment to visit or remote area distant from daily life	Managed greenspace
Aesthetical needs	As what it is	Rural people – productivity and stewardship; Urban people – 'being away' and ecological values	Neat, intention and care; increasingly notion of urban ecological values
Importance of forb performance to the public	Not important	Not important	Extremely important

2.2 Synthesising forb rich grassland communities

2.2.1 Competition as the main factor shaping a grassland community

Competition generally refers to the negative effects on plant growth or fitness caused by the presence of neighbours, usually by reducing the availability of resources (Keddy and Cahill, 2012). This ultimately occurs between neighbouring individuals (Weiner, 1990). There are various factors determining and shaping the composition of a plant community. Light, nutrients, space and water are the major resources in the competition (Grime, 2002). Within a community, seedling density, time of emergence, growth rate and other morphological variables interact to affect the amount of resources available to each plant (Bannister, 1976; Grime, 2002). Competition can appear both between different species (interspecific competition) and between the members of the same species (intraspecific competition) (Bannister, 1976). A common measurement of competition outcome is the biomass production of an individual or population which is usually considered to reflect plant growth or fitness (Mead, 1968).

Under condition of high productivity and low disturbance, species which can capture more resources than others are defined as 'competitors' in the CSR theory (Grime, 2002). Capturing the key resources modifies the local environment and suppresses, even eliminates, the neighbours. Competitive species are usually associated with a combination of rapid growth rate, high increase in height, lateral spread and root mass etc, and high capacity of phenotypic plasticity in the established stage (Grime, 2002). In the early stage, plants which emerge first within a population also have advantages to access the resources before the competition is formed (Firbank and Watkinson, 1987). This advantage may help to develop taller seedlings in the early stage. It can be compounded over time as they intercept more light resources and accumulate more biomass for a better competitiveness later on (Tremmel and Bazzaz, 1993). Köppler and Hitchmough (2015) also argued that under productive site condition, the competition is mainly between leaves and shoots for light where water and nutrient are abundant to each individual. The 'competitors' are likely to be affected by intraspecific competition sooner where the seedling density reaches a certain level. On the other hand, the competition can be reduced under the condition of low productivity or intense disturbance (Grime, 2002). The competitive relationships between species may be changed by the relative fitness of those species.

Competition between individuals can be symmetric or asymmetric. In general, the larger individual is barely affected by the smaller one (Freckleton and Watkinson, 2001; Weiner, 1990). In an even-aged communities, competition is not necessarily asymmetric from the beginning. It initially takes place between the roots for water and nutrients without much interference. Then it turns into asymmetric competition when plants are large enough to shade one another (Weiner, 1990). The relevant growth rate (RGR) as well as the intrinsic size of a species may determine its ultimate foliage height in the community and this difference causes size-asymmetry (Connolly and Wayne, 1996). This can lead to a massive reduction in the light resources available to the smaller individuals. The resource removal exponentially reduces the performance of the smaller ones and even leads to elimination. Thus, light exploitation is often acknowledged as the major factor forming asymmetric competition (DeMalach *et al.*, 2016 and Schwinning and Weiner, 1998). Within a closed community, small difference in height can lead to large changes in the species biomass composition and seedling composition in successional stages (Grime, 2002). In practice, only a slight relative advantage may be sufficient to suppress a neighbour to achieve asymmetric competition (Hitchmough, 2017a).

Within a community, plants accumulating biomass as they grow causes greater inter and intra specific competition that cause the competitive elimination to diminish the number of individuals and eventually thin the seedlings down to a sustainable number. This was discovered and defined by Yoda et al. (1963) as self-thinning. Again, productive conditions in the absence of disturbance can drive these effects to take place sooner as it increases the rate of biomass accumulation without restrictions. Higher seedling density and the distribution frequency of large plants can also increase the effect (Westoby, 1981).

The interaction between the individuals within a community is not always negative. As resources become more limiting as in unproductive habitats, the stress gradient shifts the competitive interactions to become beneficial and species sometimes facilitate each other. As competition for resources increase eventually resource shortfalls occur and hence competitive effects become reduced (Maestre *et al.*, 2009). Plants can protect each other from herbivores, potentially competitors or climate extremes as well as allocate additional resources through canopy leaching, microbial and mycorrhizal network (Bannister, 1976, Grime, 2002 and Partzsch and Bachmann, 2011). However, this ‘facilitation’ mainly occurs as a minor part of the ecological process. Grime (2002) also suggests that most of the scientific results proving facilitation effects between plants are mainly derived from plants in laboratory environments which are usually more stablished and ideal, rather than the real situation. Thus, competition is still the key issue shaping a community in landscape projects (Hitchmough, 2017a).

Herbivory also plays a key role in shaping plant communities, often in conjunction with competition. Depending on timing and amount of tissue consumed, herbivores can alter the population structure of plant and re-structure the community by impacting on the seed bank and the individual life cycle (Del-Val and Crawley, 2005 and Wilby and Brown, 2001). In most cases, herbivory effects are seen to promote species richness in a community by reducing the capacity of the most rapid growing species from outcompeting the slower growing species (Del-Val and Crawley, 2005). This occurs because the most rapid growing species are often the most palatable (Cates and Orians, 1975)

Dominants and subordinates

In a community, the competitive species accumulate the advantage of growth and ‘own’ the greatest proportion of the biomass. This group of plants controls the abundance and fitness of other minor contributors within the same system. This role is defined as ‘dominance’ (Grime, 2002). The difference of ‘scale’ and ‘precision’ in resource foraging distinguishes the role between dominant and subordinate (Campbell, Grime and Mackey, 1991). The dominants tend to exploit more resources in both space and time while the subordinates can employ the small patches of resources left by the dominants to survive but remain small (Grime, 1987). As the dominants forage a much higher proportion of resource, the dominance effect tends to be severe and suppress the subordinates, especially when the soil productivity increases (Keddy, Twolan-Strutt and Shipley, 1997). When dominance increases, the species density declines in a community (Grime, 1973). Owning to light competition playing the major role in shaping the community structure (Köppler and Hitchmough, 2015), the subordinates which can survive in the longer term are likely to have above average shade tolerance. The slow growing and full sun preferring species are likely to be eliminated sooner. However, no single species is capable to completely dominate a community. Intra specific competition occurs when the dominants biomass reaches a certain level. This leads to the mortality of dominant species and leaves unused resources, especially space and light, to the subordinates (Aarssen, Schamp and Pither, 2006). Despite the dynamic shifts within a community, the dominance normally lasts in the long term. The long-term experiment at Rothamsted Experimental Station (initiated in 1856) records the persistence of grass dominance patterns and suggests that altering and stabilising the dominance pattern by constantly changing NPK levels may take up 40 years (Silvertown *et al.*, 2006). Herben *et al.*, (2003) found that the dominant grass biomass can return to the similar level within a short term after being selectively removed. Also, enhancing species diversity in a dominance situation is suggested to be an extremely slow process in grassland restorations (Walker *et al.*, 2004).

However, the dominants are not always the ‘competitors’ according to which strategy is favoured by the habitat conditions. For example, on unproductive sites, ‘stress tolerant’ species which sustain slower growing rates under limiting conditions over comparatively long periods are likely to develop large stature to dominant in a community (Grime, 2002). Low yielding species may have a competitive advantage under extreme environment conditions (Bannister, 1976). Also, altering environmental conditions for the same plant mix may favour different species to take advantage of the competition process and even shift the role between dominants and subordinates. Sooner or later a dominant or a few dominant species which are well fitted to the environment still inevitably emerge as dominance is a relative phenomenon (Hitchmough, 2017a).

There are a very large number of species that consistently occupy subordinate positions within herbaceous community as an outcome of evolution (Aarssen, Schamp and Pither, 2006 and Grime, 2002). According to Grime (2002), there are three propositions that permit the survival of subordinates;

- Exploiting similar niches associated with different dominant species;
- Exploiting the resources left from restricted development of potential dominants by environmental and biotic factors;
- Exploiting the resource left from the temporary absence of dominants.

Subordinates account for high species richness in plant communities. The survivorship of them is the key to determine the ecological value regarding species diversity, and productivity in many cases (Mariotte *et al.*, 2013). The loss of subordinates can directly and indirectly affect the community structure and succession in a longer term through the alteration of identity, recruitment and relative abundance of dominants (Grime, 2002). With the event of major climate shift or the change of management scheme, the dominant species may decline and one or a few subordinate species can become the new dominants (Grime, 2002 and Sluis, 2002).

Coexistence

The precondition for a high species coexistence is for factors to be present which limit the appearance of dominance. Nonetheless, the presence of coexistence is a competition outcome from the past. An increase in coexistence can therefore be encouraged by the effects of stress and disturbance which is associated with reduction of potential growth of dominants (Grime, 2002).

Apart from the studies of species competition, niche occupation may also explain a pattern of coexistence. The smaller species associated with less biomass but still successful, exist within the community due to the diversity in morphological difference which help them constantly explore the spatial resources left from the large individuals (Aarssen, Schamp and Pither, 2006). Tilman (1982) also suggests that coexistence is achieved by exploiting different ratios of above and below ground resources, particularly light and nitrogen. More effectively exploiting the resources below-ground with a higher ratio of root: shoot may be the strategy responding to the limiting availability of light resource (Cao and Ohkubo, 1998).

Communities with higher species coexistence are potentially more productive (Partzsch and Bachmann, 2011 and Pokorny *et al.*, 2004). Therefore, in terms of creating productive synthetic plant communities, diversity provides the capacity to fill up niches giving the community more stability with designed species and reduce invasion of undesired species (Tilman, Reich and Knops, 2006). This pattern again reveals an equilibrium between species at the 'mid or end point' that is reached after a series of competition. This assumption is theoretically feasible in ecological models however creating a meadow from a 'blank sheet' in urban landscapes is often a different issue all together. Simultaneously different size and other characteristics also speeds up asymmetric growth and competition that certain species gain dominance over the rest.

2.2.2 Forb performance as a critical issue in grassland communities

Forb performance not only contributes to landscape values from aesthetic point of view, it also has critical effects on ecological aspects. A forb rich community is therefore a culture and nature win-win solution in designed landscapes (Hitchmough, 2017a). Ecology wise, forbs are the most diverse group within a grassland system (Dickson and Busby, 2009 and Lauenroth and Adler, 2008). Forb functional groups account for the majority of richness and biomass of a grassland plant community (Pokorny *et al.*, 2004). Morphologically, forbs are more varied in stem architecture and root pattern. This diversity contributes to explore and occupy a wider range of ecological niches.

Thus, the loss of forb richness and abundance potentially results in lower biomass production and lower community resilience in response to weeds invasion and climate change (Pokorny *et al.*, 2004 and Tilman et al, 2006). Although there is little evidence that resilience to invasion is correlated with the species richness in designed meadows, utilising different canopy layers of forbs can extend the time period of vegetative ground coverage in a year which effectively reduce weeds invasion (Hitchmough, Wagner and Ahmad, 2017). Besides, a forb-rich grassland community can significantly support more pollinators as the key process for wildlife diversity (Potts *et al.*, 2009).

In practice, the focus of ecological restoration works has been shifted from degraded mining area to the agricultural lands associated with intensive cultivation histories from the mid-20th Century (Pywell *et al.*, 2003). This transformation began to require species diversity and ecological functions rather than a simple vegetation cover on sites. Agricultural grasslands originally suffered significant loss of forb species and are now commonly dominated by a few competitive perennial grass species (Pywell *et al.*, 2003). This is mostly due to NPK fertilisation which made the soil over productive. The fertility commonly results in a quick disappearance of stress tolerant forbs from the vigorous species competition, driven by perennial grasses in most of cases (Pywell *et al.*, 2002). Introducing animal grazing back to the community can theoretically benefit the forb communities but mostly lead to a temporary appearance of ruderal forbs (Bullock *et al.*, 2001). Thus, achieving forb establishment and persistence is key to the successful ecological restoration of grasslands (Dickson and Busby, 2009).

From an ecologists point of view, grass dominance is often seen as the major problem reducing forb species richness and ecological values in grassland systems particularly where associated with intensive agriculture uses (Sluis, 2002; Pywell *et al.*, 2003; Del-Val and Crawley, 2005). Increasing the ratio of biomass of forb to grass and creating forb dominant communities usually involves approaches of existing dominant grass biomass removal plus forb seeds additions (Stevenson, Bullock and Ward, 1995; Edwards and Crawley, 1999a; Sluis, 2002; Pywell *et al.*, 2003, 2007). Increased sowing density of forb seeds has also been tested to increase the likelihood of forb dominance from the beginning (Dickson and Busby, 2009). However, forb dominance has generally been found unlikely to persist after approximately three years' time. The problem of grass dominance cannot be solved with the 'one-stop' approach. In Del-Val and Crawley (2005) research, a forb dominant community was maintained by graminicide application to suppress the growth of grass for three growing years. However, when the graminicide treatment ceased, the biomass ratio of forb: grass was gradually reduced. This fate of grass dominance in grassland restoration is mostly determined by the productive soil from the outset. Persistent seed banks and vigorous growth form allow grasses to rapidly colonise and perform increasingly well with time in the absence of effective management (Pywell *et al.*, 2003; Del-Val and Crawley, 2005).

Restoring species richness requires a persistence of species richness post the cessation of 'weed' (plant removal) and 'seed' (plant addition) (Lockwood and Pimm, 1999) and thus ecological studies and grassland restoration reviews often conclude that recreation of forb rich grasslands is only technically feasible for a relatively short period of time (Walker *et al.*, 2004). In the longer term, grass competition inevitably reduces forb richness irrespective of the initial forb seedling density and establishment (Stevenson, Bullock and Ward, 1995; Dickson and Busby, 2009). Grasses in general are superior competitor that suppress forb biomass in a grassland system as the traits associated with grasses indicate the ability 1) to outcompete others and 2) to decrease the risk of grass mortality (Pywell *et al.*, 2003).

In the long-term field experiment at Rothamsted, although the species composition is dynamic within the forb or grass group with the change of annual climate, grass dominance persists and the equilibrium point between forb and grass biomass is essentially stable in the fertile grassland system (Silvertown, 1980; Storkey *et al.*, 2016). Despite the fact that forb species are able to coexist with grasses in a grassland either by recruitment from seed or by vegetative growth even under intense competition (Del-Val and Crawley, 2005; Dickson and Busby, 2009); Staab *et al.*, (2015), found that extinction of forb species diversity (>50% of forb species) in a community requires grasses to represent 95% of the total biomass production. At these grass biomasses however the forb biomass is tiny and would not be considered to be a satisfactory community for urban landscapes.

From the 21st Century, the potential ecological values of urban areas have been recognised by ecologists and landscape architects to accommodate species diverse habitats and connect the wider green corridors (Marzluff and Rodewald, 2008). This emphasises the ecological value of creating forb-rich grasslands in urban landscapes. In respect to social values in urban landscape projects, forb performance, to some extent, may be required at a higher level. A level of forb dominance, rather than 'survival' or remaining in the subordinate role, is needed for maximising visual impacts. However, forbs are rarely the dominant species in natural or semi-natural grasslands, so to turn this around in the longer term is a major ecological and technical challenge (Hitchmough et al, 2008; Pywell *et al.*, 2007).

Fortunately, urban areas can provide a different scenario which potentially supports forb rich meadows. The brownfields in urban areas often consist of unproductive 'wastes' for example subsoils, mineral aggregates and crushed building materials. This creates extraordinary opportunities to develop the forbs which have stress tolerating characters and shade tolerance/low palatability (Hitchmough, Kendle and Paraskevopoulou, 2001). There are more chances to manipulate the site conditions, for example the topography, mulch layer, and species composition from the sowing onwards. Meadows made in these environments have turned out to be successful in terms of the persistence in forb dominance and richness in a longer period (e.g. Hitchmough, 2017a). Non-native forbs are seen as inappropriate to be introduced in the rural areas, due to both ecological consideration and social acceptance of landscape character. However, these can be applied in many urban landscape projects to provide significant additional visual effects as well as equivalent, or even in some cases more persistent, ecological values (Hitchmough, 2011).

However, there are practical problems in using diversity in this way as to have diversity low productive environments are required, leading to diverse but open communities that are more vulnerable to invasion and may also make a low contribution to visual effects. Also, in a forb dominated vegetation, most of the species are winter deciduous. In a mild winter, this absence creates physical space and light availability at soil level and leads to winter and spring germinating weeds to establish (Hitchmough, 2017a). Invasion can be reduced by adding more grasses or other forbs with evergreen foliage or persist dead foliage to the community (Hitchmough, 2017a). However, this may cause grasses to outcompete forbs or other dominants changing the community dynamic.

2.2.3 What makes grass more competitive than forbs?

Dominance of grasses over forbs has been commonly observed in a wide range of productive herbaceous communities both in the wild (Pywell *et al.*, 2003; Sluis, 2002) and also in designed urban landscapes (Bjørn *et al.*, 2019; Hitchmough, 2017). Where grasses are observed to dominate, for the most cases, this was associated with low floristic diversity (Grime, Hodgson and Hunt, 1988). Outcompeting of forbs by grasses creates a problem of reduced forb density and with this reduced “flowery-ness” which could lead to lower social acceptance and ecological value. The mechanism of grass competitiveness has been widely studied (Edwards and Crawley, 1999; Hitchmough, Kendle and Paraskevopoulou, 2001; Pywell *et al.*, 2003; Bosy and Reader, 2006; Hitchmough, Paraskevopoulou and Dunnett, 2008).

Grass species tend to be superior than forb species in survivorship from the initial stage. In fertile grasslands in the UK, the number of grass seedlings recruited from soil seed banks are much higher than forb seedlings (Edwards and Crawley, 1999). Grass therefore gains numeric advantages from the outset. In the research of Hitchmough, Kendle and Paraskevopoulou (2001), the grasses could achieve a higher and earlier emergence than forbs. This potentially gives grasses more competitiveness than the forbs. This study suggested this was because grasses have medium to large size of seeds among selected stress-tolerant species (Hitchmough, Kendle and Paraskevopoulou, 2001). Bigger seeds contain larger carbohydrate reserves facilitating seedling survival under limited availability of water (Ben-Hur and Kadmon, 2015; Jurado and Westoby, 2006). However, this advantage of grass may be mitigated if large seeded forbs are introduced in the sowing mix.

In the seedling stage, grass seedlings can develop a rapid root expansion for exploitation of the substrates and above-ground shoot growth rate for resource hunting (Campbell, Grime and Mackey, 1991; Hitchmough, Kendle and Paraskevopoulou, 2001). These advantages are enhanced when the productivity increases and generate greater effects of dominance (Campbell, Grime and Mackey, 1991). These characters help grasses dominate key resource especially light and nutrients giving them superior survivorship to neighbouring forbs (Williams, Jackson and Smith, 2007; Del-Val and Crawley, 2005). Early advantage in growth has strong relationship to the plants overall competitive performance. This can further enhance the relevant growth rate of grass seedlings by suppressing the neighbours (Pywell *et al.*, 2003). This is likely to be key to performing better in seedling competition.

With established grasses, denser tillers may be the profound morphological advantages to outcompete forb species (Pywell *et al.*, 2003; Bosy and Reader, 2006). Grass shoots can close the canopy faster and form a dense layer of litter above the ground. The foliage of grasses typically persists for longer period than the most of forbs (Grime, Hodgson and Hunt, 1988; Grime, 2002; Hitchmough, 2017a). This is assumed to impose additional stress to suppress forbs development and persistence. Dense grass leaf litter can lead to a reduction in amplitude of daily fluctuation in air temperature, which accompanies a reduction in the maximum daily air temperature. Less fluctuation in air temperature can cause a low germination rate of neighbouring forbs (Bannister, 1976 and Bosy and Reader, 2006). A mechanical barrier can be formed by grass leaf mass supressing young forb seedling to penetrate to access the light (Bosy and Reader, 2006). This is due to a lot of forb species having horizontal orientation of cotyledons that makes it difficult to push upwards through the grass litter. This leads to high mortality because there is insufficient light beneath the litter for photosynthesis to match respiration (Bannister, 1976) or because higher humidity increased infection from pathogens (Facelli and Pickett, 1991). Grass litter also forms a physical barrier to prevent forb seeds from being in contact with mineral soil (Ruprecht and

Szabó, 2012). In chemical terms, biologically active leachate from grass litter can also reduce forb emergence (Werner, 2006). However, the effect may not be significant in the field (as opposed to in laboratories) due to more complex microbial activities and heavy rain reducing the concentration of the inhibitors. Finally, grass tend to have longer life span. Resources will be made available for new grass seedling establishment through the death of neighbours especially in absence of disturbance (Lauenroth and Adler, 2008).

In terms of disturbance factors, grasses are typically more tolerant of disturbance than forbs. Grass tolerates frequent defoliation by grazing or cutting because its clonal growth form and the position of their perennating buds just below or close to the soil surface (Pywell *et al.*, 2003). This makes grass recovery and canopy closure faster, decreasing the risk of grass mortality. Disturbance would especially become less effective to increase forb species diversity when the grasses are established in summer (Bullock *et al.*, 2001). Forbs gaining advantages to survive and establish from disturbance are normally short term whereas grasses soon seize back the dominance role (Walker *et al.*, 2004; Del-Val and Crawley, 2005).

Moreover, grass is less palatable to molluscs. Where grasses are dominant molluscs densities are often high due to the structure of the grass canopy (Edwards and Crawley, 1999; del-Val and Crawley, 2005). Grass often contain silica in their leaves and are often avoided by molluscs leading to differential grazing pressure on forbs. In addition, because of the timing of attack on forbs usually occurs at the seedling stage rather than mature plants, molluscs cause high forb seedling mortality. This encourages transition from a forb-dominated community to a grass-dominated community (Wilby and Brown, 2001).

Once a perennial grass dominated community is established, seedlings of the same species are soon be established in the surrounding gaps or even replace the existing seedlings groups (Edwards and Crawley, 1999; Dickson and Busby, 2009). Hitchmough (2009) found that 200mm wide gaps was not sufficient to stop grass invasion to eliminate forbs previously established by planting, including forbs native to the area in grassland. This pattern is unlikely to be turned around unless a large scale disturbance to the dominant grass in combination with reintroducing desired species are applied to a grassland community (Edwards and Crawley, 1999; Sluis, 2002; Walker *et al.*, 2004).

Most of the studies above strongly support the superior competitiveness of grass species. The outcome is often based on complex effects from large scale measurements with long term observations. There are a few previous researches carried out at smaller scales which may provide another insight. A study (Dwyer, 1958) carried out on a grassland site where were relatively undisturbed (i.e. free from grazing or other unnatural disturbances for many years) in Kansas, the US found that the spontaneous forbs (*Ambrosia psilostachya, Aster ericoides, Aster oblongifolius, Solidaga mollis* and *Solidago rigida*) with rhizomatous growth form can suppress dominance grass (*Andropogon gerardi, Andropogon scoparius* and *Bouteloua curtipendula*) through direct contact with each other in the surface four inches of soil Rhizomatous forbs tend to be more vigorous in growth and competitive for space. This causes a biomass reduction in grass. Besides, the forbs with taproots (*Echinacea angustifolia, Gutierrezia sarothrae, Petalostemon candidum, Petalostemon purpurea Psoralea tenuiflora* and *Schrankia uncinate*) can coexist well with grasses by utilising moisture and nutrients below the root system of grasses. In addition, Tremmel and Bazzaz (1993) suggested that the forbs which produce a larger leaf area than adjacent grasses can impose more light stress to the neighbouring grass individuals and to be more competitive than grasses. However, both studies were short term (one growing season) and small scaled, which reduces capacity to reflect on the overall community dynamic and development between forbs and grasses. Also, the studies merely cropped and captured a small picture of the interaction between some established plants. This excluded the factors for example, responsible for the difference in emergence and seedling growth rate between forb and grass species.

2.2.4 Designing a semi-natural grassland community in urban landscapes

From the late 20th Century in the UK, management funds for public parks have been dramatically reduced due to a series of political changes and financial crisis (Layton-Jones, 2016). Traditional plantings appeared less affordable to maintain to high quality in greenspace, in terms of resistance to invasion and regeneration. Such plantings usually require extreme intensity of horticultural disturbance (maintenance) to prolong their performance, especially in the longer term. Monoculture planting may need complete replacements when the designed species reaches the limit of life span or experiences any extreme climate. Landscape managers then started to seek out an alternative way maximising the effectiveness of management resources to provide both ecological and social values (Kendle, Rose and Oikawa, 2005). This idea brought back the focus onto planting models in nature which are supposed to be self-regulated but also visually attractive as communities (Köppler and Hitchmough, 2015). Naturalistic planting especially the meadow-like vegetation, which derived from natural plant communities synthesising between ecology and design, have systematically been studied and developed as a tool for urban landscape architecture from the early 21st Century (Hitchmough and Dunnett, 2004). This planting style has been increasingly fashionable in Western Europe level (Köppler and Hitchmough, 2015).

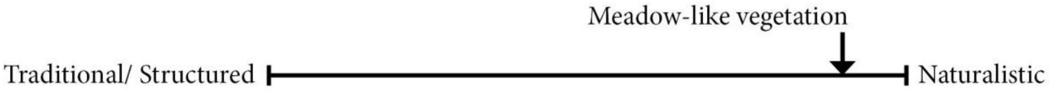


Figure 2.1 Meadow-like vegetation are strongly naturalistic in philosophy and the appearance is highly closed to the grassland communities in nature

Naturalistic communities involve design with species-rich and multi-layer plantings. Forbs in particular are planted in random arrangements that mimic natural plant community (Refer to Figure 2.1). They can involve a diversity of species which occupy different niches and will establish and reproduce whenever the environmental condition is favourable (Hitchmough and Dunnett, 2004; and Bjørn, Weiner and Ørgaard, 2016). ‘Attractiveness’ is less dependent upon the productivity of individual plants but the entire vegetation (Hitchmough, Kendle and Paraskevopoulou, 2003). Management approaches are often easier and more time efficient compared with conventional horticultural vegetation since the activities are applied to a whole community rather than individual plants (Hitchmough, 2017a). The consideration of environmental sustainability is equally important and can be comprehensively achieved by a naturalistic community by reducing carbon emission of management equipment and supporting a wider range of wildlife (Hitchmough and Dunnett, 2004; Özgüner, Kendle and Bisgrove, 2007).

The mechanism of coexistence of many plant species in a diverse community of this type is not yet fully understood, however the major challenge is the persistence of forb species diversity and richness.

Can a forb rich community be persistent in urban landscape?

The increasing notion of ecological and biodiversity values in urban area brought up the idea of 'stability'. It mostly implies an equilibrium point between community properties which can underpin persistent productivity, that is resistant to invasion and can return to the original state after disturbance (Köppler and Hitchmough, 2015). A diversity of precedent planting projects have proved that highly weed resistant communities can be achieved with a complex structure on relatively high productivity soils (Hitchmough and Wagner, 2013; Hitchmough, Wagner and Ahmad, 2017; Bjørn *et al.*, 2019).

However, within these communities, full persistence of the original species, as designed may never exist, species continue to change spontaneously, as is the nature of vegetation process. Without regular disturbance, competition in combination with herbivory, inevitably takes place leading to aggregation of biomass from planted or incoming colonising individuals . Any minor change to resource allocation may contribute to asymmetric growth and change the envisaged pattern at the local level. This process spontaneously drives the community structure. A stable community has always been an illusion (Bjørn, Weiner and Ørgaard, 2016 and Hitchmough, 2017).

Species richness diminishes with passage of time

With the passage of time, the loss of species richness in designed communities is mainly due to competitive elimination, namely self-thinning, and weed competition in some cases (e.g. Hitchmough, De La Fleur and Findlay, 2004). The original seedling number is often reduced by half after about three growing years in a carefully designed herbaceous mix (e.g. Hitchmough and Wagner, 2013; Bjørn *et al.*, 2019). The effect of initial species composition and spatial distribution would be diminished (Bjørn, Weiner and Ørgaard, 2016; Bjørn *et al.*, 2019). Symmetrical growth across each species does not occur in a meadow community since the difference in individual stature is accentuated by the variety of key traits (for example relevant growth rate) responding to the site conditions and neighbouring individuals. This again inevitably leads to the pattern that vigorous species, playing the dominant role and representing the major coverage and biomass, create subordinates irrespective to the mix originally designed (Hitchmough, Wagner and Ahmad, 2017). Light competition still plays the key role to the self-thinning and loss of diversity in meadow-like communities in urban landscapes (Vojtech, Turnbull and Hector, 2007; Hitchmough and Wagner, 2013).

Disturbance can be effective but re-induction of new plants is needed

Appropriate management approaches can extend the duration of forb richness in a community. The key is to balance out resource allocation and keep the survivorship as high as possible since most of the seedlings remain subordinate. The increase of light access at the soil level for re-emerging seedlings is usually achieved in meadow-like communities by standing biomass removal. The community can return to the same productivity level within the year and the disturbance is often required annually (Bjørn *et al.*, 2019). The timing of application is critical to the pattern of dynamics. Early cutback, for example in August, can disadvantage the tall species but increase the survival of short subordinates (Hitchmough and Wagner, 2013). However, reduction of vegetative cover in summer can make the community vulnerable to invasion (Bjørn, Weiner and Ørgaard, 2016). On the contrast, a late cutback maximises potential photosynthesis and hence dry weight of individual, providing their foliage is arranged in space and time in such a way as to intercept light (to a community). Species with tall or widely spreading foliage are likely to benefit from this (Hitchmough, 2009). This mechanism leads to small species elimination sooner but can reduce the establishment of invaders.

However, the evidence that disturbance effectively improves species diversity in a community is weak or even negative (Köppler and Hitchmough, 2015). In a designed community, the effect of recruitment from self-seeding within the system usually appears too low to change the dynamic, especially where there is no seed rain from the outside into the meadow in most cases. Over-sowing or transplanting combining with disturbance therefore is needed for preserving a colourful and species-rich forb community in the longer term (Dickson and Busby, 2009 and Bjørn, Weiner and Ørgaard, 2016). In semi-natural meadows diversity in a given patch is significantly maintained by seed rain from outside the patch (Hitchmough, Paraskevopoulou and Dunnett, 2008; Hitchmough, 2017a).The alternative is to accept a lower level of species diversity which is stable at a given level of management and productivity.

The trend for diversity to diminish being the norm as part of the ecological process. Completely self- regulated and self-regenerating communities do not really exist in either the natural world or in the landscapes of cities. It is however possible to maintain forb rich communities for long periods of time, to deliver politically attractive ecological experiences and develop social values in urban landscapes. This brings a key question; can design process extend the persistence of forb richness, at least for a relatively long term?

2.2.5 To what extent, can the starting point change community performance through design in the longer term?

There are three main factors playing the key roles in designing a semi-natural community; resource availability, plant community composition and disturbance (Cascorbi, 2007). These factors affect the interactive relationship between the individuals as well as community pattern in both short term and long term. In general, the strategy is to achieve a longer coexistence by limiting the proportion of the large growing potential dominants to reduce competition with small growing species. Design approaches usually involves setting up the initial seedling densities and manipulating soil fertility and moisture status of the soil (Hitchmough, De La Fleur and Findlay, 2004; Hitchmough, 2017a). Relatively high community productivity reduces seedling invasion from the outside and quickly achieves a substantial visual impact which is critical to gain public support. However, in general, taxonomic diversity decreases with increasing density of potential dominants or soil productivity (Keddy, Twolan-Strutt and Shipley, 1997; Hautier, Vojtech and Hector, 2018). Thus, the best possible solution to maintain equilibrium is to maintain a relatively productive but diverse community, however this is always something conundrum especially in communities created by sowing.

Theoretically, the effect of initial seedling composition diminishes after few years but the accumulated effects of survival, growth and regeneration strategies become dominant to shape the community appearance in the long term (Bjørn, Weiner and Ørgaard, 2016). It is inevitable that seedling mortality and recruitment flux as a part of this ecological process. However, designing the starting points can substantially affect the dynamic within a community. In an urban semi-natural community, the design process can be involved from a very beginning prior to site construction and sowing, and then into the longer term. This provides the opportunity to design the initial species composition, and control the level of resource availability, through for example by different types of mulch layers, or soils. This is likely to affect the biomass distribution and the pattern of coexistence for a longer period which may also affect the pattern of light competition and delay the impact of dominance effects.

First year survival is important to represent the species composition in a relatively longer term (Lauenroth and Adler, 2008). A higher number of surviving seedlings allows more opportunities for community regeneration in respect to the scope of management and potential climate change. Maximising the initial ratio of biomass of desired species to weeds can be a good indicator to a persistent community in the longer term (Hitchmough and Fleur, 2006). Combining sensible management (usually low resource input required such as cutting), designed communities can effectively satisfy both ecological and social needs, and sustain this in the long term. Examples of projects in practice that have achieve this include; The Merton Borders at University of Oxford Botanic Garden, Sheffield Botanical Gardens Prairie and large scale project of Queen Elizabeth Olympic Park UK Native Wildflower Meadows (Hitchmough, 2017).

Species selection

'Species' is as an extremely important variable in the design decision and significantly affects the community structure and behaviour (Hitchmough, Wagner and Ahmad, 2017; Bjørn *et al.*, 2019). Some guidance for species selection has been proposed to extend forb-rich persistence by practitioners. Firstly, selecting species with similar key traits, for example growth rate, can increase stability of mixed plantings (Köppler and Hitchmough, 2015).

Secondly, species that fit within the ecological conditions of the habitat, but are as different as possible in their niches within the habitat would tend to coexist better (Bjørn, Weiner and Ørgaard, 2016). Thirdly, species performing equally well under low resource-requiring maintenance and having similar response to major disturbance such as mowing are advantageous to persist (Bjørn, Weiner and Ørgaard, 2016). Finally, the capacity for survival more likely determines the persistence of a species than the reproduction ability in a perennial community (Bjørn *et al.*, 2019). Pywell *et al.*, (2003) also found that higher vegetative competitiveness is more important than abilities of succession and colonisation for a species to persist in the long term in a semi-natural community.

Sowing times, watering intervals and initial establishment

Sowing time and irrigation concurrent with the germination window are critical to enhance initial percentage emergence. Emergence is higher when species whose seed has a chilling or after-ripening are subjected to periods of low temperature chilling, a common phenomenon in temperature species (Baskin and Baskin, 2014). Thus autumn or winter sowing are advantageous for these species (Hitchmough, De La Fleur and Findlay, 2004). Irrigation and in particular the frequency of irrigation increases percentage emergence even for the species from dry habitats (Fay and Schultz, 2009; Hitchmough and Wagner, 2013; Hitchmough, 2017a). Many meadows are undermined from the outset in the absence of temporary irrigation, as forb numbers are often very low, especially if dry weather is experienced during the emergence window. Higher seedling numbers close the ground sooner, making a community more resistant to weed invasion in the first year by reducing invader establishment (Hitchmough, 2017a). Higher seedling numbers will of course potentially lead to greater self-thinning, so a density balance has to be struck. A community, in terms of seedling survivals and growth, will be harder to control from the second year as an entirely different dynamic will appear. The carbohydrate accumulated from the growth of first year boosts growth from the next spring exponentially increasing the size of individual plants (Hitchmough, 2017a). The interactive effects thus also exponentially increase. As individual size gets larger from the second growing year, disturbance such as plant removal aiming to reduce the density of dominant species may leave large gaps on soil surface for weeds invasion.

Are there any design approaches that can reduce dominant effects and encourage the persistence of subordinate forbs and high species richness?

Longer persistence of forb richness in a designed community requires a higher forb survival but lower presence of dominants, which can be either forb or grass species. In principle, greater forb persistence can be achieved by;

1) Increasing forb diversity and survivals in the first year;
2) Limiting grass presence to a threshold biomass;
3) Reducing the number and density of potentially dominant forbs/grasses;

4) Limiting resource availability to discourage the growth of potential dominants.

The first three of these “levers” are largely associated with conditions in the first growing season and are strongly related to community design and establishment protocols. The fourth lever is important in both the first year and into the distant future.

Increasing sowing rate was traditionally used in agriculture as a non-herbicidal way to reduce weed establishment by a quick closure of plant canopies when dealing with monocultures (Andrew and Storkey, 2017). The same mechanism has also been applied in grassland restoration works and urban planting projects. Both were shown to have significant effects on the establishment of desired species with a reduction of weed invasion (Stevenson, Bullock and Ward, 1995; Hitchmough and Fleur, 2006). The initial sowing rate and emergence is positively related and even almost be linear which at least effectively enhance the forb richness in the short term, however, the biomass ceases to increase after reaching a certain threshold (Hitchmough, De La Fleur and Findlay, 2004; Hitchmough, Paraskevopoulou and Dunnett, 2008). Higher seedling density leads to the competition taking place sooner and more intensely leading to lower seedling survival (Hitchmough and Fleur, 2006; Hitchmough, Wagner and Ahmad, 2017). A lower sowing density is likely to aid sown seedlings surviving into the second year providing the site has few undesired plants weedy colonists which will compete with the sown species (Stevenson, Bullock and Ward, 1995; Hitchmough and Fleur, 2006; Scotton, 2019). The use of sand and other sowing mulches is effective in reducing emergence of these weedy species from the soil seed bank (Hitchmough and Fleur, 2006; Hitchmough, 2017a).

Within a community, grass species are likely to outcompete the forbs sooner when the sowing rate increases in both. Higher seedling ratios of forbs to grasses potentially increase forb survivals and biomass in the short term but not the long term (Dickson and Busby, 2009). However, a high sowing density of shade tolerant forbs can be effective to retain forb abundance to coexist with grasses even reduce the grass biomass in the longer term (Hitchmough, Paraskevopoulou and Dunnett, 2008). The efficiency of increasing forb sowing density depends on various factors including site productivity, species in sowing mix and the existing seedling composition.

In the study of Hitchmough, Kendle and Paraskevopoulou (2001), a series of stress tolerant forbs were shown to be able to emerge and survive under severe moisture stresses. Imposing moisture stress can perhaps reduce stress-tolerant forbs mortality from competitors of both competitive grasses and forbs. This process may be indirect; the restricted growth of potential competitors would allow more light resource allocating to slow growing species (Keddy, Twolan-Strutt and Shipley, 1997). However, there is as yet no evidence to support if this model (Hitchmough, Kendle and Paraskevopoulou, 2001) can maintain forb richness in longer term. Another consideration is that a certain level of productivity is often needed to deliver the ecological and visual values in urban landscapes. The level of moisture stress is critical; extreme moisture stress drastically reduces the germination even of those species from dry habitats (Fay and Schultz, 2009; Hitchmough, 2017a). So initial irrigation is really helpful, but then variability in soil moisture once emergence has occurred is also useful to young seedling survival (Fay and Schultz, 2009), some of which show lower survivorship at constant and high irrigation frequency (Hitchmough, 2010). This requires critical decision making on the irrigation scheme to balance emergence percentage and seedling survival.

Designing to have higher forb survival leads to intense competition and may increase the dominance effect. However, altering the availability of key resources can perhaps adjust the threshold of dominance and improve the persistence of forb richness with the same species composition. To our knowledge, there are few tests of grass competition with forb species in which design is used to manipulate the starting point to provide a community that contains some grass biomass to perform critical function roles whilst retaining forb richness. Also, to date there are no clear guidelines on critical values of biomass of dominant species, or density in relation to subordinate forb persistence in the literature.

Foliage height and plant architecture in relation to light competition

Developing leaves above the neighbours is the primary strategy to win in light competition (Grime, 2002). Species with greater canopy height have a disproportionate competitive advantage (Vojtech, Turnbull and Hector, 2007). However, the dynamic can be more complicated depending on the site conditions and species composition. As a relative effect, the advantage in height can be accumulated from the beginning for example early germination or fast relevant growth rate at the seeding stage (Weiner, 1990; Tremmel and Bazzaz, 1993). Both advantages can help to intercept more light and shade their neighbours, restricting their capacity to achieve their potential height. Taller seedlings tend to be more competitive in the later stage (Tremmel and Bazzaz, 1993). Late emerging species (either as seedlings or adult plants) tend to be more likely outcompeted (Hitchmough, 2009).

Where a group of plants have similarities in the key traits of for example growth rate and habitat conditions, plant architecture can affect the effectiveness of light interception (Hitchmough, 2009; Ford, 2014). This difference causes asymmetric competition for light and is the major factor leading to a dominance and low diversity in a community under productive conditions (Vojtech, Turnbull and Hector, 2007). These factors are very important in forb's potential ability to survive under grass competition (Hitchmough, 2009).

Plants that allocate more growth to structural stems than leaves are likely to be most competitive for light (Westoby, 1981). This growth form also helps leaves to penetrate through the dense vegetation canopy to be superior to the neighbours and is referred to as shoot thrust (Grime, 2002). Moreover, the strength of light interception depends on the gross leaf area. Species with large but fewer leaves may not be as competitive for light as species with many small leaves (Tremmel and Bazzaz, 1993). These latter morphologies may leave fewer light gaps to the neighbouring plants.

Another key issue is leaf arrangement. Plants with upright stems or long petioles which allow leaves to be held in sunlight tend to be success in the competition whereas shade intolerant species with low or basal foliage are likely to be eliminated sooner under productive conditions (Hitchmough, 2009). In addition, the later species tend to gain more benefits from disturbance (Bullock *et al.*, 2001). Finally, leaf angle and the spatial distribution of leaf biomass where towards the sunlight can also optimise the light competitiveness (Ford, 2014).

2.3 The use of Inner Mongolian grassland species to create meadow-like grassland communities

Inner Mongolia is found between 53°N 121°E to 40°N 100°E, spanning approximately 1,200,000 km^2 as one of the most Northerly parts of China and supports a large grassland flora. There is increasing interest, in parallel with interest in meadow like vegetation in the UK in using elements of this vegetation in designed environments.

Steppes in the Mongolian region comprise various ecological gradients with distinctive planting habitats representing the largest grassland biome in the region (Kang *et al.*, 2007; Chen *et al.*, 2016). The temperate steppes efficiently service ecological functions of the region and support diverse species of plants and wildlife (Ni, 2003; Kang *et al.*, 2007; Ariuntsetseg and Boldgiv, 2009). Since the very first scientific study of the Mongolian flora commenced in 1870s present research on Mongolian grassland plants and ecosystem is quite advanced (Kang *et al.*, 2007). There are considerable gradient changes in altitude (140m – 1,700m), annual mean temperature (0.7 °C – 6 °C), and annual mean precipitation (400 – 600mm) from the east to the west (Ni, 2003; Wang, 2004) leading to a diversity of grassland habitats (Ni, 2003).

Most of the research on these grassland ecosystems of the Mongolian region are mostly single-discipline/perspective based (Kang *et al.*, 2007; Shinoda, Nachinshonhor and Nemoto, 2010). There is a lack of research and understanding towards utilising this ecosystem as a vegetation type for urban areas for example. In Inner Mongolia, both landscape or horticulture academics have not shown much interest in trying to do this. In the UK, many Mongolian forb species are commercially available and have been widely cultivated as ornamental planting such as *Thermopsis lanceolata*, *Delphinium grandiflorum* and *Kalimeris incisa* etc. (Royal Horticultural Society, 2020). However, there is limited data of plant traits available, such as the Ellenberg's indicator values, CSR values, and leaf fresh/ dry weight from precedent studies and the Plant Trait Database (TRY).

As previously mentioned, a key factor in successfully establishing a species rich community is to control the biomass trade-off between grasses and forbs. Interestingly, many natural Mongolian grassland communities show that some forbs can coexist with productive grasses very well (Liu, Jiang and Duan, 2015). This character might be worth investigating and utilised in flower rich urban meadows.

An interesting link between the meadow/ steppe flora of Inner Mongolia, and the meadows of the UK is that many species are distributed from Western Europe to Asia and are present in both countries. These cross over species provide an opportunity to test hypotheses as to how nativeness influences plant survival in grasslands and in particularly when subject to competition from grasses. For example, would 'shared' species always show a better compatibility or competitiveness that the 'exotic' species in the 'native' habitats? Would 'fitness' to certain habitat also show a stronger ability to coexist with grass?

Chapter 3 Methodology

3.1 Experiment site and materials

3.1.1 Experiment site

The experiment was located at Manor Top (53°37′82″N 1°43′51″W), Sheffield. The experiment field sat on a west facing slope previously used for agriculture and cultivation with a fertile topsoil (as explained in 3.1.3). The field surroundings were cultivated meadow planting mixes and shrub mass. A 1.5m wide gap was maintained around the experiment to reduce weedy ingress. Ground preparation work took place in the middle of March 2017 including spraying herbicide to controlled perennial grasses (twice) and cultivating the ground. Experiment plots and sand substrate were installed from 8th May, and the seed mixes were sown in the 23rd May 2017.

3.1.2 Species selection

In order to test whether non-native to the UK species were more sensitive to competition than the species that were originally distributed across the UK and Eurasia (especially Inner Mongolia, China) in the long term study, 29 forb species were selected and divided into two categories; 1) species with distributions in Western Europe and Inner Mongolia, and 2) species restricted to Inner Mongolia. The selection criteria were based on (see also Table B in Appendices):

- Foliage canopy height (3 categories, essential to the experiment because of the significance of light competition between forbs and grasses, in addition to layers related to visual appreciation)
- Growth rate/potential productivity
- Adequate fitness for cultivation in Northern England
- Attractiveness and landscape value
- Capacity to be germinated from spring sowing in the UK
- Low palatability to molluscan herbivores
- The commercial availability of seeds

The selected species are listed as follow:

Forb species	**Low foliage canopy ≤ 300mm**	**Medium 300-600mm**	**Tall foliage canopy ≥ 600mm**
Shared distribution between Inner Mongolia and W. Europe	*Anemone sylvestris* *Galium verum* *Potentilla rupestris* *Pulsatilla vulgaris*	*Achillea millefolium* *Stachys officinalis* *Campanula glomerata* *Origanum vulgare*	*Echinops ritro* *Geranium pratense* *Sanguisorba officinalis* *Thalictrum aquilegifolium*

	Veronica teucrium	*Polemonium caeruleum*	*Veronica longifolia*
Inner Mongolian distribution only	*Dracocephalum rupestre* *Dracocephalum ruyschiana* *Thalictrum petaloideum* *Thermospsis lanceolata* *Veronica incana*	*Campanula punctata* *Delphinium grandiflorum* *Kalimeris incisa* *Platycodon grandiflorus* *Scutellaria baicalensis*	*Aconitum carmichaelii* *Angelica sylvatica* *Echinops sphaerocephalus* *Patrinia scabiosifolia*

The species seeds were purchased mainly from *Jelitto Perennial Seed. Campanula glomerata* was purchased from *Emorsgate Seeds* and *Deschampsia cespitosa* ‘Barcampsia’ was from *Barenbrug.*

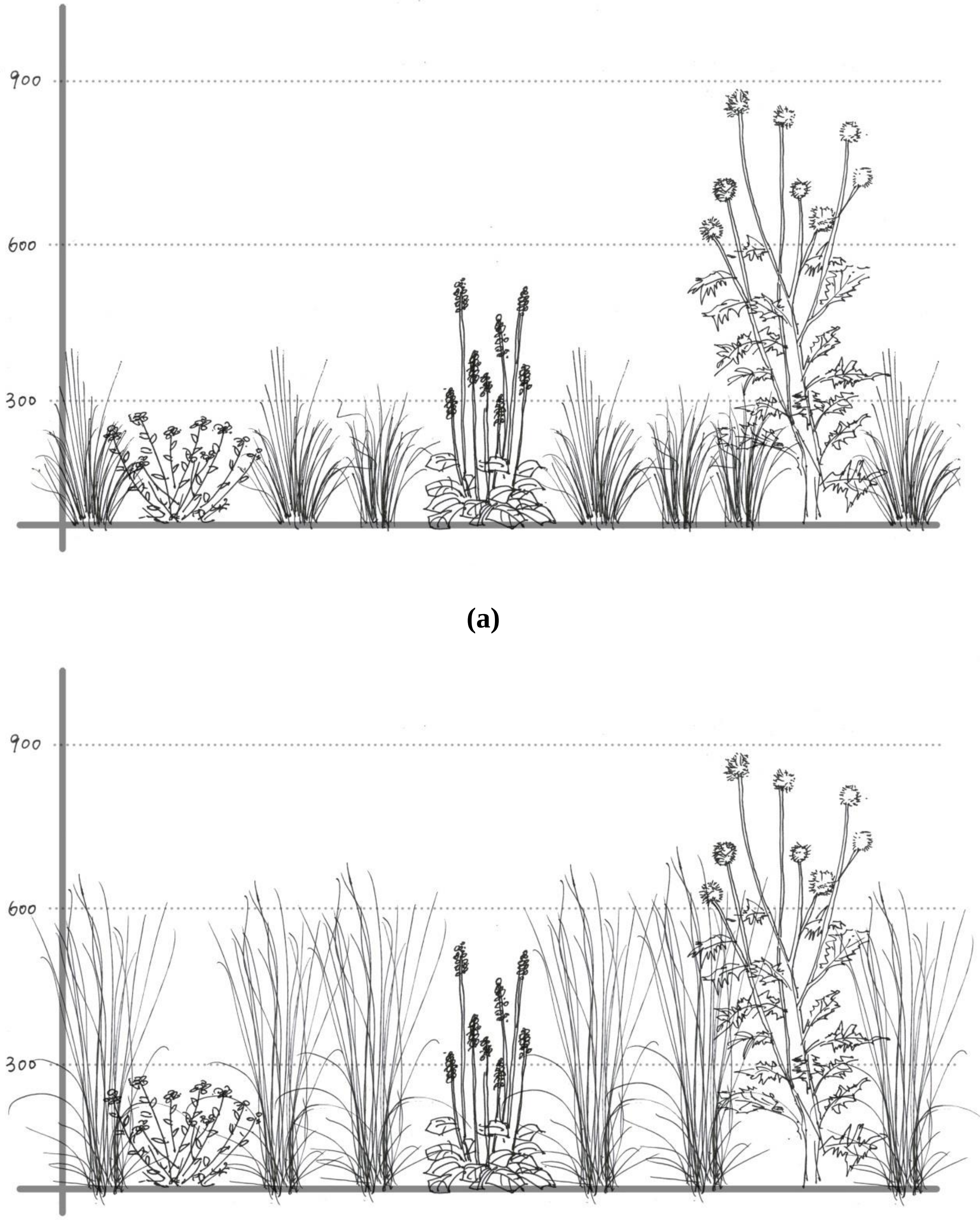
Because hard seed coats reduce water absorbing for *Thermospsis lanceolata*, seeds of this species were scarified by sandpapers to enhance the germination rate. Also, Gibberellic acid (GA3) treatment (1,000 ppm) were applied to the dormant seeds of *Stachys officinalis, Aconitum carmichaelii* and *Angelica sylvatica* for better germination percentage in a spring sowing. *Campanula punctata*, *Geranium pratense* and *Pulsatilla vulgaris* were purchased as commercially GA3 treated seed to ensure adequate emergence from the spring sowing.

To avoid having another species as a variable, two distinctive genotypes of the grass *Deschampsia cespitosa* were used (small and less competitive and larger and more competitive). This species was chosen because it is:

- Widely distributed across the world (including Western Europe and Inner Mongolia)
- Relatively attractive and persistent
- Readily availability

Deschampsia cespitosa are widespread in moist grasslands in both Western Europe and Inner Mongolia, and commonly used in designed landscape plant community. This species forms tussock and remain structurally intact and reasonably attractive post flowering, which is important in designed landscapes. Most C3 grasses of both European and Inner Mongolian grasslands do not possess the characteristics. These two genotypes were *Deschampsia cespitosa ‘*Pixie Fountain’ (foliage canopy approximately 300mm tall) and *Deschampsia cespitosa* ‘Barcampsia’ (foliage canopy approximately 700mm tall)

Figure 3.1 below shows an indicative forb height category in relation with both ‘low’ (a) and ‘tall’ (b) grasses in designed grassland communities.



(a)

(b)

Figure 3.1 Forb heights in relation with the ‘short’ and ‘tall’ grasses

3.1.3 Mulch layer selection and existing topsoil

Grit Sand (MKM Building Supplies) was used as the experimental substrate. This was used to help provide a relatively high percentage seedling emergence, whilst controlling subsequent growth rate (by lowering moisture and nutrient levels) and suppressing weed immergence from the underlying soil. The particle diameter was approximately 0.05mm – 2mm and pH 6.5, i.e. neutral.

The sands have higher water supplying capacity comparing with rubbles and low resistance to root penetration. This allowed seeds to have sufficient uptake of water and emerged seedlings to access sub-surface moisture during drier periods (Hitchmough, Kendle and Paraskevopoulou, 2001). In addition, it was essentially weed seed free, reducing subsequent weed establishment. Sand is often use in meadow sowing in practice for these reasons.

The soil type of existing topsoil beneath the sand layer was categorised as clay loam. The soil pH was 5.5. Soil nutrient analysis to contextualise the site was not available, however as biomass harvesting was a key part of the research methodology this provided a much more meaningful measure of potential productivity. The level of productivity as an integrated indicator of key site conditions including nutrients, moisture, and annual climate etc. can give the most critical context to field experiments on meadow communities. This site supported approximately 1008g and 1218g standing biomass on average per unit area (800 x 800mm excluding the 200mm edges) in 2018 and 2019 (at the time point the plant roots effectively penetrated through the sands). This level of above ground biomass (i.e. > 900g/m2) production corresponds to upper levels possible in non-wetland sites in both UK (Qi *et al.*, 2018) and Inner Mongolia (Ni, 2004), and is much higher than the productivity values associated with species rich meadow communities (typically 300-600g/m^2). To put this into context, general experience in gardens with intensive fertilisation history usually support 900 – 1,200g/m^2 of garden plant biomass (Hitchmough, 2017a). Highly productive soils are common in urban areas and the site represented the worst case scenario in terms of likely competitive dominance in meadow making in urban landscapes.

Figure 3.2 Grit sand (left) and clay loam topsoil (right) samples

3.2 Experiment layout

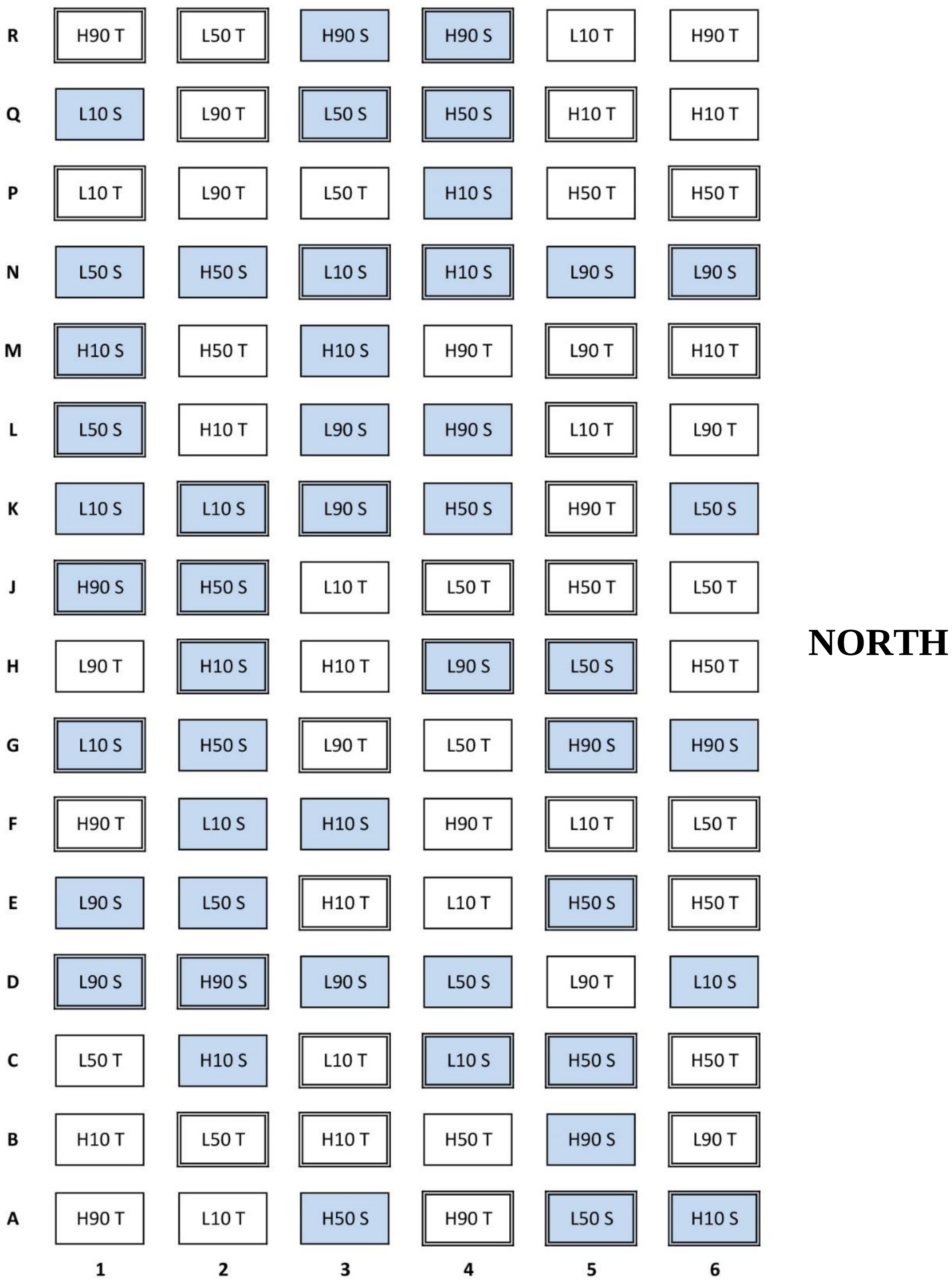
The hypotheses underpinning the experiment involved the following positions:

1) Higher density sowing lead to competitive elimination sooner.
2) Forb total standing biomass and species richness will be enhanced when the initial ratio of forbs: grasses increased.
3) Lower soil productivity will keep forb species diversity for a longer period.
4) Taller forb and grass species will be more competitive.

A factorial experiment was designed that involves following factors to test the hypotheses above:

- **2 seedling densities (as initial seedling numbers)**: Low: 500 and High 1,000 (forbs + grasses) seedlings/m^2
- **3 ratios of forbs to grasses (as initial seedling numbers)**: 1: 9, 1: 1 and 9: 1
- **2 soil productivities (mainly moisture):** higher (75mm sand mulch layer) and lower (150mm sand mulch layer)
- **2 grass foliage heights:** Short: *Deschampsia cespitosa* 'Pixie Fountain' (300mm) and Tall: *Deschampsia cespitosa* 'Barcampsia' (700mm)

These four factors led to have 24 treatment combinations, with 4 replicates of each treatment combination making a total of 96 plots (Figure 3.2). After considering the visual effects and meadow community structure, the seedling targets represented a ratio in terms of species height of that species low: medium: tall of approximately 4:2:1. The ratio of tall forbs to other forb categories was higher than usual meadow sowing mix for urban landscapes, where the difference of the target number of seedlings of low canopy v tall canopy species is much greater than four folds, as this is a key means of reducing competitive elimination of small species by the tallest species. The 4:2:1 was a necessary evil to ensure that 1) sufficient tall forb seedling emerged in the low sowing rate treatment and 2) to ensure sufficient light competition. It was however anticipated to lead to greater elimination of subordinate species.



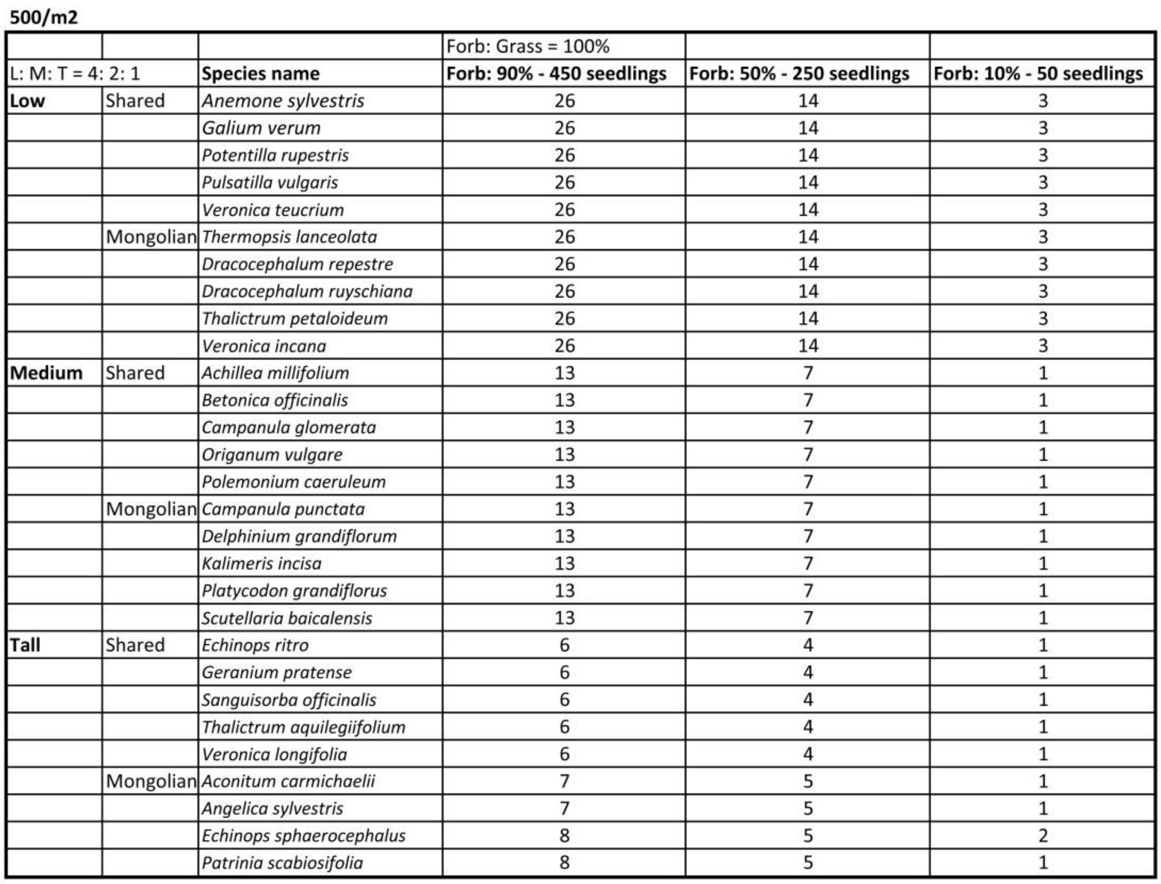
	1	2	3	4	5	6
R	H90 T	L50 T	H90 S	H90 S	L10 T	H90 T
Q	L10 S	L90 T	L50 S	H50 S	H10 T	H10 T
P	L10 T	L90 T	L50 T	H10 S	H50 T	H50 T
N	L50 S	H50 S	L10 S	H10 S	L90 S	L90 S
M	H10 S	H50 T	H10 S	H90 T	L90 T	H10 T
L	L50 S	H10 T	L90 S	H90 S	L10 T	L90 T
K	L10 S	L10 S	L90 S	H50 S	H90 T	L50 S
J	H90 S	H50 S	L10 T	L50 T	H50 T	L50 T
H	L90 T	H10 S	H10 T	L90 S	L50 S	H50 T
G	L10 S	H50 S	L90 T	L50 T	H90 S	H90 S
F	H90 T	L10 S	H10 S	H90 T	L10 T	L50 T
E	L90 S	L50 S	H10 T	L10 T	H50 S	H50 T
D	L90 S	H90 S	L90 S	L50 S	L90 T	L10 S
C	L50 T	H10 S	L10 T	L10 S	H50 S	H50 T
B	H10 T	L50 T	H10 T	H50 T	H90 S	L90 T
A	H90 T	L10 T	H50 S	H90 T	L50 S	H10 S

Figure 3.3 Experimental plot arrangement plan on site (H/ L = high/low sowing density; 10/ 50/ 90 = sowing ratio of forb: grass of 1: 9/ 1: 1/ 9: 1; S/ T = short/ tall grass; single plot outline denotes 75mm mulch depth; double line plot outline denotes 150mm sand substrate)

Target seedling numbers for both forbs and grasses for each treatment were designed in order to achieve a target number of seedlings (Table 3.1 and 3.2). The amount of seed sown into each of the 96 plots was calculated for each species in each plot based on seeds/gram (Jelitto, 2017), emergence rates (Hitchmough, 2010; Jelitto, 2017) and estimated percentage emergence (Hitchmough, 2017a).All the mixes differed in seed density and the ratio between forb and grass but not species richness.

The dimension of site is about 11 x 24m. 96 1x1m randomised timber plot placed evenly within the area leaving 1m (vertical) and 0.5m (horizontal) gaps between the plots. From the 5th to 16th May 2017, all the plots stapled with damp proof course inside to prevent timber rotten. Ground matting then applied to prevent weeds growth on the gaps. About 25 tons of sharp sands filled into all the plots according to designed depths as the substrate. Seed mixes sowing commenced and were raked into sands on 23rd until 25th May (as shown in Figure 3.3 – 3.6).

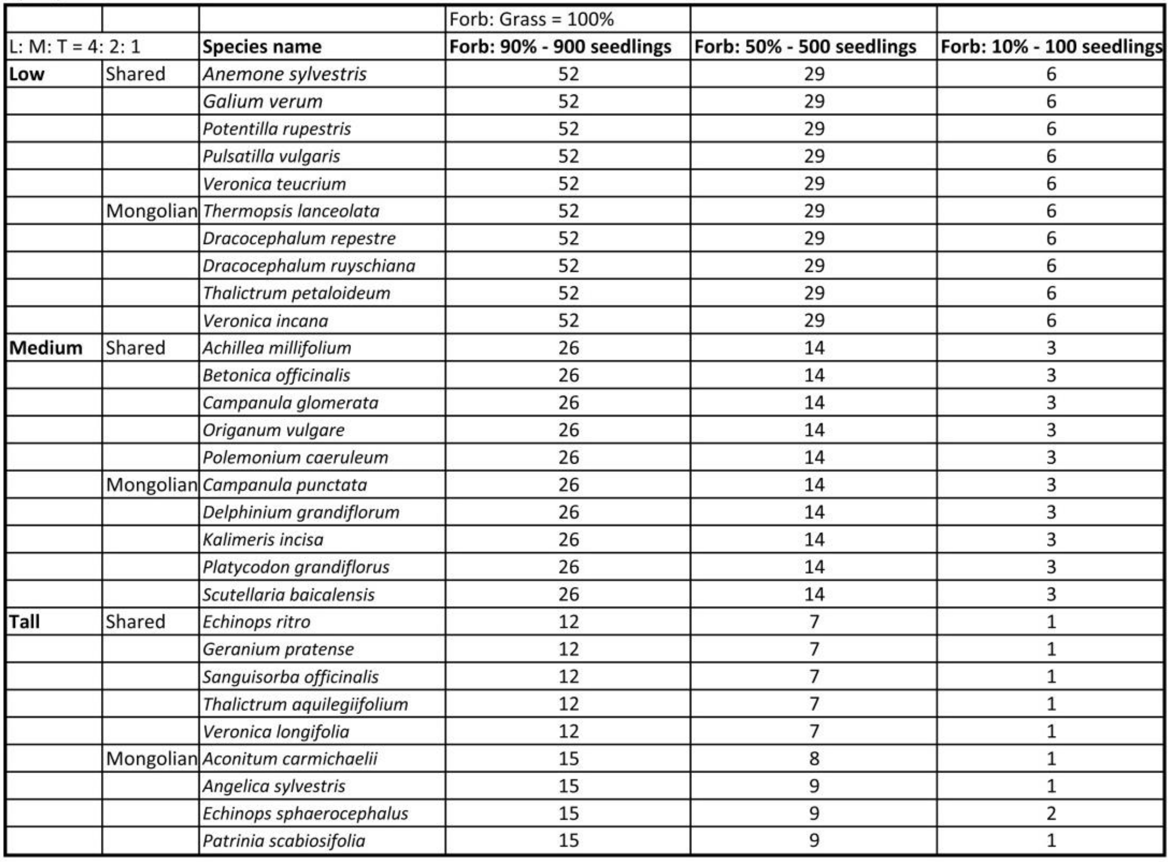
Table 3.1 Target seedling numbers for forb and grass at two sowing densities x three sowing ratios of forb: grass. Sowing densities of 500/m2 (low) and 1,000/m2 refer to target emergence numbers of 500 forb + grass seedlings; and 1,000 forb + grass seedlings. Sowing ratios of forb: grass refer to target emergence ratio of forb: grass of 9: 1 (90% forb – e.g. 450 forb seedling in the low sowing density; 900 forb seedling in the high sowing density, respectively); 1: 1 (50% forb) and 1: 9 (10% forb)



500/m2

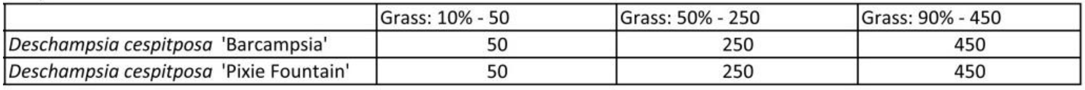
			Forb: Grass = 100%		
L: M: T = 4: 2: 1		**Species name**	**Forb: 90% - 450 seedlings**	**Forb: 50% - 250 seedlings**	**Forb: 10% - 50 seedlings**
Low	Shared	*Anemone sylvestris*	26	14	3
		Galium verum	26	14	3
		Potentilla rupestris	26	14	3
		Pulsatilla vulgaris	26	14	3
		Veronica teucrium	26	14	3
	Mongolian	*Thermopsis lanceolata*	26	14	3
		Dracocephalum repestre	26	14	3
		Dracocephalum ruyschiana	26	14	3
		Thalictrum petaloideum	26	14	3
		Veronica incana	26	14	3
Medium	Shared	*Achillea millifolium*	13	7	1
		Betonica officinalis	13	7	1
		Campanula glomerata	13	7	1
		Origanum vulgare	13	7	1
		Polemonium caeruleum	13	7	1
	Mongolian	*Campanula punctata*	13	7	1
		Delphinium grandiflorum	13	7	1
		Kalimeris incisa	13	7	1
		Platycodon grandiflorus	13	7	1
		Scutellaria baicalensis	13	7	1
Tall	Shared	*Echinops ritro*	6	4	1
		Geranium pratense	6	4	1
		Sanguisorba officinalis	6	4	1
		Thalictrum aquilegiifolium	6	4	1
		Veronica longifolia	6	4	1
	Mongolian	*Aconitum carmichaelii*	7	5	1
		Angelica sylvestris	7	5	1
		Echinops sphaerocephalus	8	5	2
		Patrinia scabiosifolia	8	5	1

1,000/m2



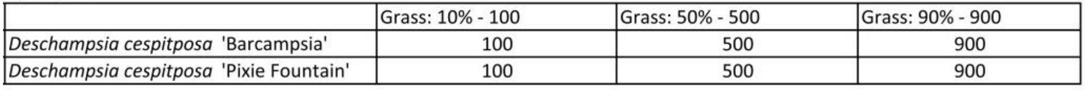
			Forb: Grass = 100%		
L: M: T = 4: 2: 1		**Species name**	**Forb: 90% - 900 seedlings**	**Forb: 50% - 500 seedlings**	**Forb: 10% - 100 seedlings**
Low	Shared	*Anemone sylvestris*	52	29	6
		Galium verum	52	29	6
		Potentilla rupestris	52	29	6
		Pulsatilla vulgaris	52	29	6
		Veronica teucrium	52	29	6
	Mongolian	*Thermopsis lanceolata*	52	29	6
		Dracocephalum repestre	52	29	6
		Dracocephalum ruyschiana	52	29	6
		Thalictrum petaloideum	52	29	6
		Veronica incana	52	29	6
Medium	Shared	*Achillea millifolium*	26	14	3
		Betonica officinalis	26	14	3
		Campanula glomerata	26	14	3
		Origanum vulgare	26	14	3
		Polemonium caeruleum	26	14	3
	Mongolian	*Campanula punctata*	26	14	3
		Delphinium grandiflorum	26	14	3
		Kalimeris incisa	26	14	3
		Platycodon grandiflorus	26	14	3
		Scutellaria baicalensis	26	14	3
Tall	Shared	*Echinops ritro*	12	7	1
		Geranium pratense	12	7	1
		Sanguisorba officinalis	12	7	1
		Thalictrum aquilegiifolium	12	7	1
		Veronica longifolia	12	7	1
	Mongolian	*Aconitum carmichaelii*	15	8	1
		Angelica sylvestris	15	9	1
		Echinops sphaerocephalus	15	9	2
		Patrinia scabiosifolia	15	9	1

500/m2



	Grass: 10% - 50	Grass: 50% - 250	Grass: 90% - 450
Deschampsia cespitposa 'Barcampsia'	50	250	450
Deschampsia cespitposa 'Pixie Fountain'	50	250	450

1,000/m2



	Grass: 10% - 100	Grass: 50% - 500	Grass: 90% - 900
Deschampsia cespitposa 'Barcampsia'	100	500	900
Deschampsia cespitposa 'Pixie Fountain'	100	500	900

Table 3.2 Estimated % of emergence for designed forb and grass species (the seeds weighing process and calculation shown in the Table C in Appendices)

Forb species	Estimated % of emergence
Anemone sylvestris	21
Galium verum	30
Potentilla rupestris	25
Pulsatilla vulgaris	27
Veronica teucrium	30
Thermopsis lanceolata	30
Dracocephalum repestre	30

Dracocephalum ruyschiana	17
Thalictrum petaloideum	26
Veronica incana	12
Achillea millefolium	30
Campanula glomerata	3
Origanum vulgare	5
Polemonium caeruleum	31
Stachys officinalis	13
Campanula punctata	4
Delphinium grandiflorum	20
Kalimeris incisa	30
Platycodon grandiflorus	33
Scutellaria baicalensis	21
Echinops ritro	21
Geranium pratense	20
Sanguisorba officinalis	24
Thalictrum aquilegifolium	13
Veronica longifolia	7
Aconitum carmichaelii	8
Angelica sylvestris	20
Echinops sphaerocephalus	20
Patrinia scabiosifolia	34

Grass species	Estimated % of emergence
Deschampsia cespitposa 'Barcampisa'	10
Deschampsia cespitposa 'Pixie Fountain'	25

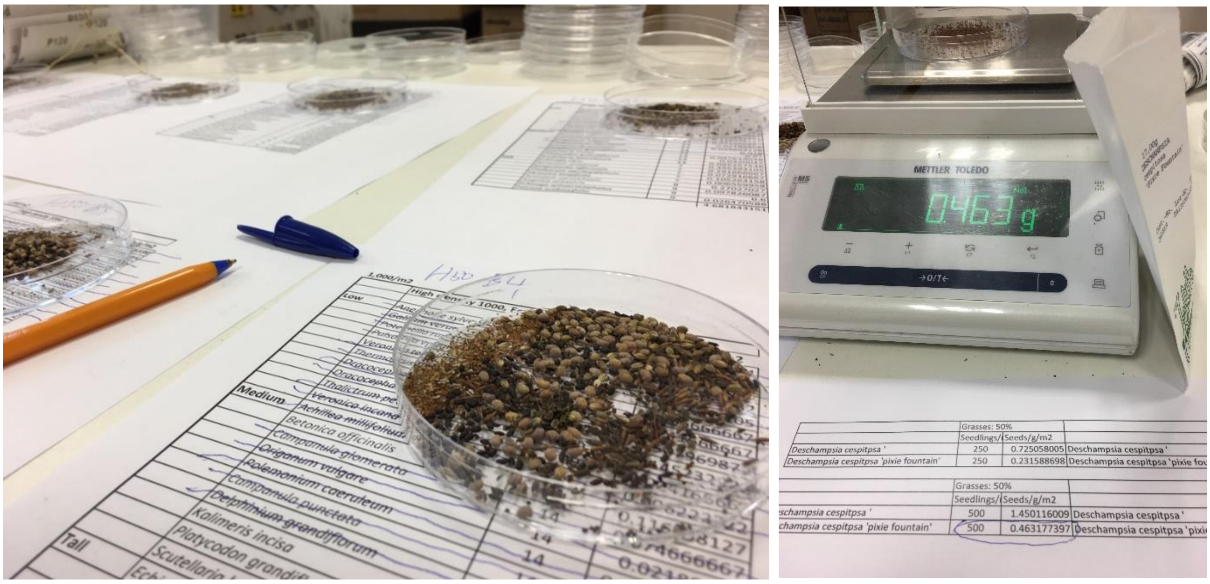


Figure 3.4 Seed mix weighing. Each species was weighed for 96 plots individually according to designed target seedling densities. This was to minimise the error from outset (pictures taken on 29th April 2017)

Figure 3.5 Frames to hold in the 75 and 150mm layers of sand mulch were made of recycled timber panels. Due to limited availability of labour resources, March and April 2017 were spent building the timber frames; the inside was lined with plastic sheeting matting to reduce wood rot in the longer term

Figure 3.6 Experiment plot set up in mid-May 2017. Approximately 27 tons of sands were filled into the plots as the mulch layer. Wheelbarrows were used when the tractor was not available

Figure 3.7 The sand mulch was raked and firmed to be prepared for sowing

3.3 Management and adjustment for the field experiment

In May 2017, hessians were stretched over each top of timber bed immediately after sowing (as shown in Figure 3.7). This created approximately 50% more shady area to keep the temperature lower and moisture post irrigation/rain to boost germination in summer. Plots were irrigated every 2 days in the absence of rain with a soft rain lance plot by plot. In late June, hessians were replaced by wire meshes to prevent fox digging when it seemed the sufficient germination had been achieved (as shown in Figure 3.8). The irrigation routine was also adjusted to once per week but stopped when soil was observed still wet. Since May and June in 2018 were extraordinarily hot and dry, manual watering reapplied every week (3 times in total) in absence of rain to reduce seedling elimination.

Figure 3.8 Hessian was stretched overall the top of each plot to retain moisture and maintain lower temperature (picture was taken on the 23rd May 2017)


Figure 3.9 In late June 2017 (approximately 5 weeks after sowing), hessians were replaced by wire mesh when emergence had generally achieved the expected level

To maximise initial species diversity in this long term study, species identified as fast growing in early summer, in this condition, heavily suppressing surrounding seedlings were selectively disadvantaged to prevent the excessive dominance and elimination in the early stage. The largest and fastest seedlings (mainly *Achillea millefolium*, *Echinops ritro, Echinops sphaerocephalus*, *Geranium pratense* and *Deschampsia cespitosa*) were cut back approximately every 10 days from the 2nd August 2017 to simulate mowing, a practice that is often applied to newly sown meadows on real life projects. Seedling numbers of *Achillea millefolium, Echinops ritro* and *Echinops sphaerocephalus* were then thinned down by manual removal to designed numbers (as shown in Table 3.1) for each plot from 5th to 13th August 2017. Grasses were also selectively thinned to achieve the designed density.

The density of the tall grass (*Deschampsia cespitosa* 'Barcampsia') was lower than expectation in July 2017. To redeem this, new seeds were purchased (these proved to have better emergence rate in parallel seedling tray tests) and re-sown accordingly on the 8th October 2017. In early June 2018, additional *D. cespitosa* 'Barcampsia' seedlings (previously grown in seedling trays, refer to Figure 3.9) were transplanted into the plot, where their density was still low, to achieve as uniform establishment before harvest as possible. However, mostly due to the extreme weather and light competition of tall forbs, both the tall grass sowing and transplanting did not achieve comparable density to that of the short grass by the time for data collection in August 2018. Relatively uniformed coverage between tall and short grass had been achieve by spring 2019.

Figure 3.10 New *Deschampsia cespitosa* 'Barcampsia' seeds were sown in seedling trays to germinate and then were transplanted into seedling pots in mid-May 2018 prior to transplanting into the experimental plots in June 2018

Aconitum carmichaelii, *Angelica sylvestris* and *Stachys officinalis*, which were separately cultivated in seed trays, were transplanted into the plots according to designed seedling numbers in late November 2017. However, due to insufficient seedlings of *Aconitum carmichaelii* and *Angelica sylvestris* in trays, seedling density of the two species were reduced from the designed seedling number. One *Aconitum carmichaelii* was planted in each plot and *Angelica sylvestris* had numbers of 1 (in low density x 10% forb and high density x 10% forb), 2 (in low density x 50% forb) , 3 (low density x 90% forb and high density x 90% forb) and 5 (in high density x 90% forb).

The 1m wide zone around the experiment was sprayed by herbicide (glyphosate) twice a year during the growing seasons to minimize weedy seeds blowing into the experiment plots. Hand removing potentially tall ruderal weeds in plots was undertaken whilst small from mid-

June 2017 to May 2018. In late June 2018, grass seed heads were manually stripped off and removed before seed shedding to avoid contamination of other plots.

Cutback to about 20mm above the ground with a petrol hedge trimmer (Figure 3.10) (followed by removal of the cut material) was undertaken at the beginning of each growing season in late February. Biomass was cutback to the same level in August 2018 and 2019 for biomass data collection, with each species sorted and its biomass placed in coded bags. This was undertaken to reflect standard meadow management in practice.

Figure 3.11 Petrol hedge trimmer was used to cutback in each February and hand snippers were used for biomass data collection in each August

3.4 Data collection

All the data were collected from an 800 x 800mm quadrat placed in the centre of each plot to minimise the edge effect (as shown in Figure 3.11). Three types of data were collected;

1) Forb seedling number present in each plot in September 2017, April 2018 and April 2019
2) Biomass weight of each species per plot in August 2018 and August 2019
3) Forb, grass and bare ground cover values in each plot in October 2017, May 2018 and May 2019

Seedling counting took about 6 weeks from early September in 2017. In April 2018 and 2019, as the author became more familiar with forb seedlings and the counting method, the counting reduced to 3 weeks.

The actual number of seedlings that emerged in summer 2017 in relation to the target numbers was not known, as seedlings were not counted till large enough to be confident about identification in September 2017. By then it is assumed that the numbers counted represented maximum emergence minus seedling mortality through competition and other sources.

Thalictrum spp. and *Veronica spp.* seedling numbers were not counted in the first year due to difficulties of identification. Also, all the *Campanula spp.* were counted as *Campanula punctata* due to difficulties and the lack of reliably distinctiveness between very small *Campanula spp.* seedlings within the quadrats. Evidence from parallel pot studies did not show emergence of *Campanula glomerata*, suggesting this species was poorly represented on the plots. This suggests the seeds quality was likely to have been low. A decision was made not to count grass seedlings due to insufficient time availability.

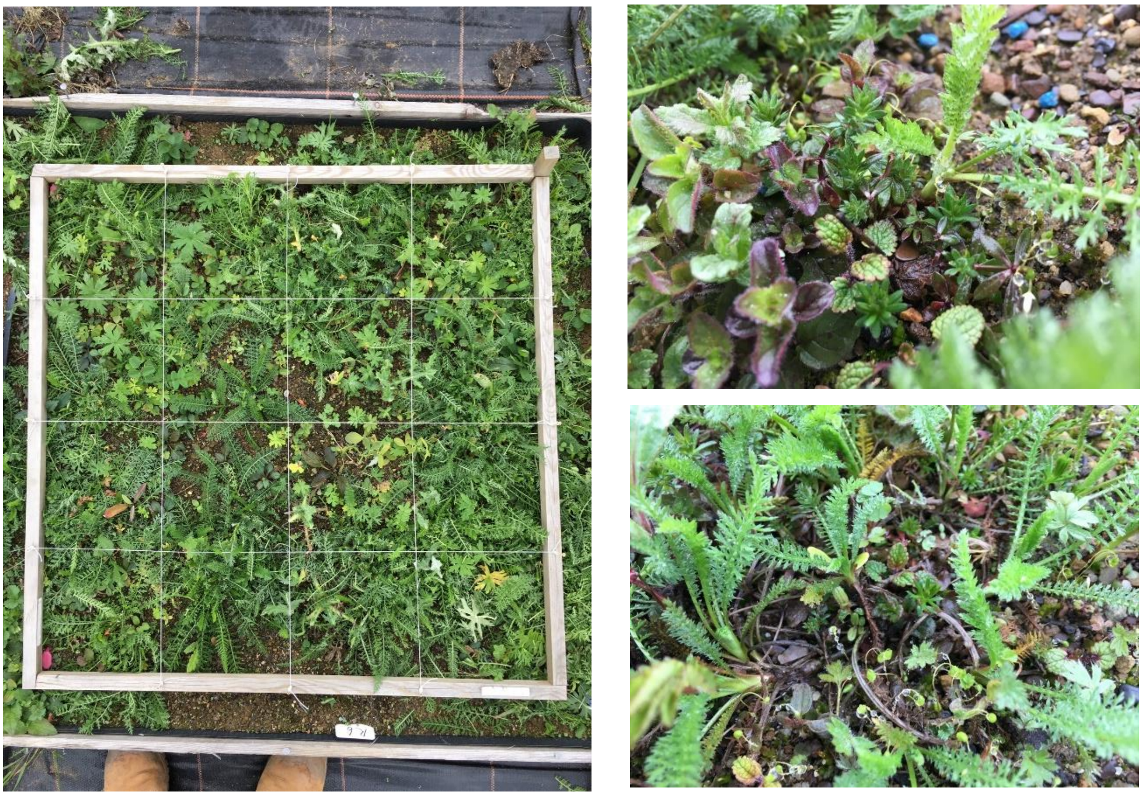


Figure 3.12 An 800 x 800mm timber quadrat was used for forb seedling counting. Plastic pegs (the red points) marked the corners of the quadrat to ensure data collection within the same permanent area in the longer term (picture on the left taken on the 3rd October 2017). Large numbers of tiny seedlings were counted within each quadrat where the forb sowing density was high (pictures on the right taken on the 12th April 2018)

Biomass harvest started in late July and took about 4 weeks in 2018 and 3 weeks in 2019 (Figure 3.12). The plant matter was cut and put into the coded paper bags for each species per plot (Figure 3.13). All the bags were stored in a dry polytunnel to allow preliminary air drying. From mid-September, the bags were dried in a drying cabinet at 75°C for 24 hours left in the room humidity and temperature for another 24 hours before weighing to achieve some consistency between weighings as weight increased quickly in the first few hours when moisture from the air was absorbed.

Figure 3.13 Dominant species biomass (mainly *Deschampsia cespitosa*, *Achillea millefolium* and *Echinops sphaerocephalus*) were collected in individual potato sacks on site. The subordinate species were brought back together and separate into individual envelopes in the lab (pictures taken in August 2018)

Figure 3.14 A corner of plant sample storage. The sample bags were later put into drying cabinet for dehydration for weighting

The data analysis shown in later chapters will mainly interpret the results from 2018 as the starting point and results in 2019 as the end point. Results in 2017 are used as a reference in some cases.

Ground level solar radiation level (watt/m^2) was measured with the Delta-T SunScan (Figure 3.14) in June and July 2019. This was used as a supporting evidence to quantify light competition of tall species with the small species at the lower level.

Figure 3.15 Delta T SunScan (on the left). Measuring solar radiation level underneath dominants canopies on the 26th June 2018

Table 3.3 Species in categories of unmanipulated, manipulated/ removed and added by transplanting in winter

Species unmanipulated (17 species)	*Anemone sylvestris* *Campanula glomerata* *Campanula punctata* *Delphinium grandiflorum* *Dracocephalum rupestre* *Dracocephalum ruyschiana* *Galium verum* *Geranium pratense* *Kalimeris incisa* *Origanum vulgare* *Patrinia scabiosifolia* *Platycodon grandiflorus* *Polemonium caeruleum* *Potentilla rupestris* *Pulsatilla vulgaris* *Sanguisorba officinalis* *Scutellaria baicalensis* *Thermospsis lanceolata* *Thalictrum aquilegifolium* *Thalictrum petaloideum*

	Veronica incana *Veronica longifolia* *Veronica teucrium*
Species manipulated/ removed (3 species)	*Achillea millefolium* *Echinops ritro* *Echinops sphaerocephalus*
Species transplanted in winter (3 species)	*Aconitum carmichaelii* *Angelica sylvatica* *Stachys officinalis*

3.5 Statistical analysis

The statistical tests were undertaken with SPSS version 26. Generalized Estimating Equations (the GEEs) were applied to build the 2 level factorial models (i.e. the model tests all possible 2-factor interactive combination). Sowing density, ratio of forb: grass, heights of grass and depths of sand substrate were treated as the factors (Subject Variables). ‘Year’ (i.e. to represent the data difference between 2018 and 2019) was treated as Within-subject Variable. Sequential Sidak correction was applied for the comparison of estimated means to obtain the significance levels.

Consulting with the statistician Dr. Jean Russell at the University Research Support Team, the accumulated biomass data and cover value data in each plot were analysed with the Linear model type within the GEEs, where the tests were valid regarding the standardised residuals and the data size. Seedling number data were treated as ‘counts’ and was analysed with Poisson Loglinear type models.

To meet the assumption of data distribution and validate the tests, other model types within the GEEs were applied and data were transformed to optimise the normality. Due to the different intrinsic size of species there was a need to standardize the scores. To test the difference of forb biomass between the treatments or years, the mean and standard deviation for each species were calculated as a best guess at the normative behaviour of the species. To standardize the raw scores then the following equation was used to obtain the z-score;

$$Z_{\text{biomass}} = (\text{Sample biomass} - \text{Mean}_{\text{biomass/ species}}) / \text{Standard Deviation}_{\text{biomass/ species}}$$

This reduced the direct effect of the factor species but still allowed assessment of interaction with respect to species.

Chapter 4 Effects of time

4.1 Introduction

The idea of community 'persistence' and 'stability' is attractive to both ecologists and landscape practitioners as this may represent continuous ecological and cultural services in a system. In urban context, retaining a high level of forb diversity and richness are important. Fully self-sustained communities do not exist. Relative persistence may occur in the longer term but as a result of competition and herbivory involves a great deal of change in species and biomass composition.

This field experiment took place between late-May 2017 to the mid-August 2019. The first three growing years were expected to have the most shifts in seedling numbers and biomass abundance (Hitchmough and Fleur, 2006; Hitchmough and Wagner, 2013). Grassland communities can be predicted to be more stabilised in the later years in the absence of major events of management changes or climate shifts. This type of pattern has also been constantly shown in ecological studies (Stevenson, Bullock and Ward, 1995; Bjørn *et al.*, 2019). Grass competition often leads to significant community dominance across these time periods (Del-Val and Crawley, 2005).

With the context of these dynamics, this chapter will explore how the community changed over the time period until summer 2019 and test the likelihood of a forb dominant community being retained in the longer term.

4.2 Results

4.2.1 Forb seedling number, forb and grass biomass and forb, grass and bare ground cover value in 2018 and 2019

The overall multi-factorial statistical model (as shown in Table 4.1) tested the level of significance of each designed factor (included 'Year') and all possible 2-level treatment combinations that impacted on the collected data including forb seedling number, forb and grass biomass, and cover values.

Each designed factor significantly affected the number of forb seedlings. Also, grass biomass was sensitive to these design approaches with the only exception of the sowing density treatment (p=0.091). However, forb biomass appeared less sensitive to most of the treatments but the sowing ratio of forb: grass treatment (p=0.000) which involved extremely different starting point (i.e. nine times difference in the initial sowing ratio of forb: to grass) was highly significant.

The treatments generally affected the pattern of forb, grass and bare ground coverage in spring. However, depth of sand substrate appeared insignificant on the forb cover value (p=0.158), sowing density appeared insignificant in terms of grass cover value (p=0.481) and forb: grass ratio was insignificant in terms of bare ground cover value (p=0.904).

'Year' significantly affected all categories of data. This showed the significant difference in the development of designed community between 2018 and 2019.

There was no significant effect for most of the 2-level interactions. The significant effects were 'Sowing density x height of grass' on forb and bare ground cover values (p=0.007 and p=0.041), 'forb: grass ratio x height of grass' on forb seedling number (p=0.001) and 'forb: grass ratio x depth of sand substrate' on grass biomass (p=0.004), grass cover value (p=0.005) and bare ground coverage (p=0.023).

Table 4.1 The 2-level model effects of designed factors on forb seedling no., biomass and cover values in 2018 and 2019 (*p ≤ 0.05; **p < 0.01, ***p < 0.001 and ns=not significant)

Factor	df	Forb seedling no. Sig. level	Forb biomass Sig. level	Grass biomass Sig. level	Forb cover value Sig. level	Grass cover value Sig. level	Bare ground cover value Sig. level
Sowing density	1	0.000 ***	0.232 ns	0.091 ns	0.005 **	0.481 ns	0.001 **
Forb: grass ratio	2	0.000 ***	0.000 ***	0.000 ***	0.000 ***	0.000 ***	0.904 ns
Height of grass	1	0.000 ***	0.164 ns	0.000 ***	0.030 *	0.001 **	0.003 **
Depth of sand substrate	1	0.000 ***	0.098 ns	0.004 **	0.158 ns	0.000 ***	0.000 ***
Year	1	0.000 ***	0.000 ***	0.000 ***	0.000 ***	0.000 ***	0.000 ***
Sowing density*Forb: grass ratio	2	0.147 ns	0.911 ns	0.753 ns	0.777 ns	0.782 ns	0.478 ns
Sowing density*Height of grass	1	0.552 ns	0.383 ns	0.594 ns	0.007 **	0.208 ns	0.041 *
Sowing density*Depth of sand substrate	1	0.437 ns	0.526 ns	0.675 ns	0.862 ns	0.782 ns	0.743 ns
Sowing density*Year	1	0.042 *	0.011 **	0.397 ns	0.031 *	0.039 *	0.000 ***
Forb: grass ratio*Height of grass	2	0.001**	0.460 ns	0.417 ns	0.640 ns	0.289 ns	0.312 ns
Forb: grass ratio*Depth of sand substrate	2	0.180 ns	0.142 ns	0.004 **	0.183 ns	0.005 **	0.023 *
Forb: grass ratio*Year	2	0.000 ***	0.164 ns	0.003 **	0.334 ns	0.135 ns	0.784 ns
Height of grass*Depth of sand substrate	1	0.056 ns	0.700 ns	0.127 ns	0.858 ns	0.615 ns	0.439 ns
Height of grass*Year	1	0.891 ns	0.000 ***	0.000 ***	0.000 ***	0.000 ***	0.001 **
Depth of sand substrate*Year	1	0.922 ns	0.075 ns	0.000 ***	0.002 **	0.026 *	0.000 ***

4.2.2 Effect of time on forb seedling numbers, forb and grass biomass and cover values

As shown in Figure 4.1, between April 2018 and April 2019, about 44% of forb seedlings (about 100 forb seedlings) decreased (p=0.000) on average in each plot. Also, forb biomass had a significant increase (p=0.000) of about 393.78g (55.94% increase) in every plot while grass biomass had a significant decrease (p=0.000) of 183.43g (60.43%) (as shown in Figure 4.2). The total biomass productivity increased from 1007.55g to 1217.89g (20.88%) per plot which was also statistically significant (p=0.000).

The number of forb seedlings decreased predominantly because of competitive elimination. However, forbs produced more abundant biomass in August 2019 and probably supressed the growth of grass.

Coincident to the change of biomass between 2018 and 2019, forb cover value had a significant (p=0.000) increase from about 40% to 70% while grass cover value decreased from 35% to 27% (p=0.000) in spring (as shown in Figure 4.3). The community establishment over the year supported more vegetative coverage in spring and made a significant reduction on bare ground coverage (p=0.000).

i) Forb seedling numbers present per plot

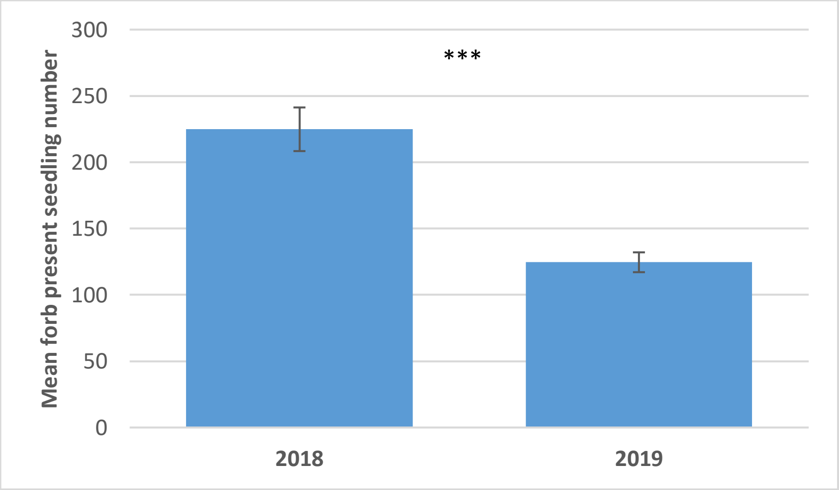


Figure 4.1 The mean forb seedling numbers present per plot in April 2018 and April 2019 (*p ≤ 0.05; **p < 0.01, ***p < 0.001 and ns=not significant. Error bar = 2 Standard Errors)

ii) Forb and grass biomass per plot

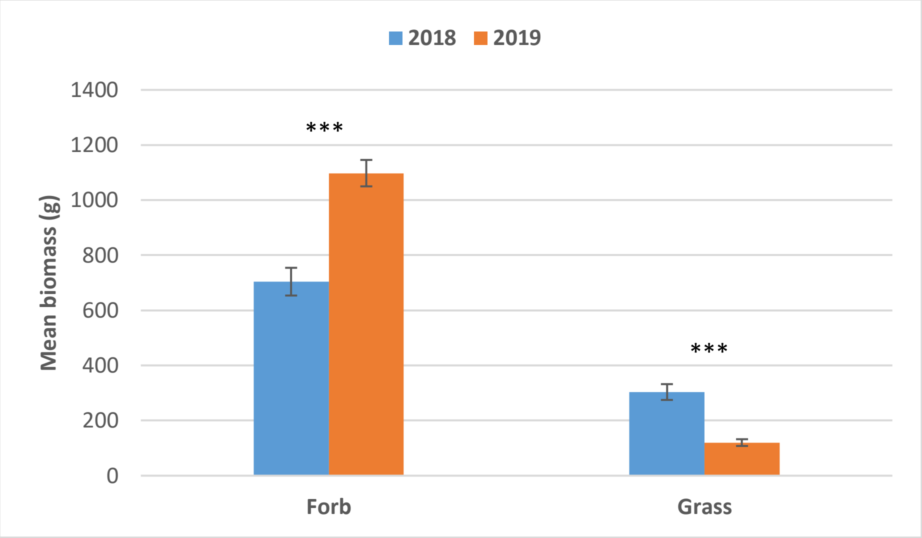


Figure 4.2 The mean forb and grass biomass (g) per plot in August 2018 and August 2019 (*p ≤ 0.05; **p < 0.01, ***p < 0.001 and ns=not significant. Error bar = 2 Standard Errors)

iii) Forb, grass and bare ground cover value

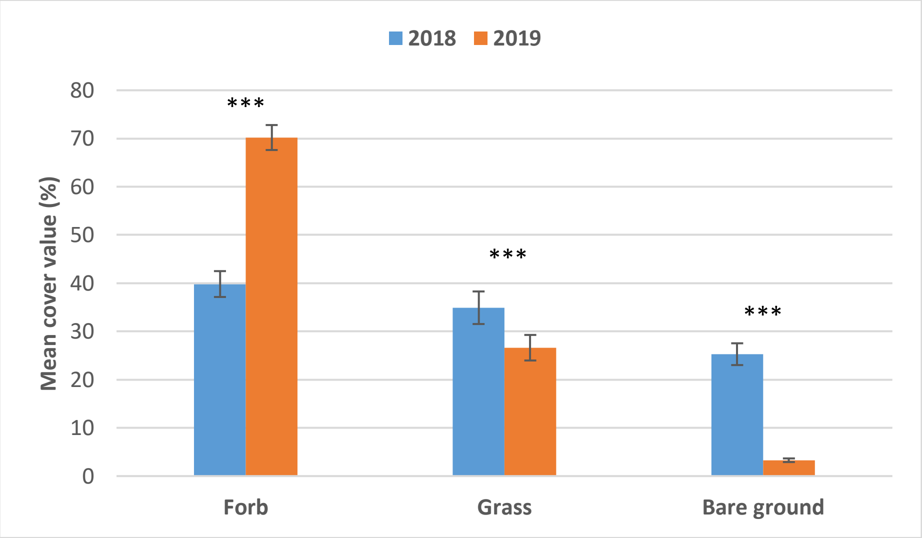


Figure 4.3 The mean forb, grass and bare ground cover value (%) per plot in May 2018 and May 2019 (*p ≤ 0.05; **p < 0.01, ***p < 0.001 and ns=not significant. Error bar = 2 Standard Errors)

4.2.3 Effect of time on dominant forbs (*Achillea millefolium* and *Echinops sphaerocephalus*) and subordinate forb biomass

As the most dominant forb in terms of biomass were *Achillea millefolium and Echinops sphaerocephalus* taking more than 90% of the total forb biomass in both 2018 and 2019, they were separated from the rest of the 26 subordinate forb species for analysis. This, especially in the results below, would reveal whether the dominants or subordinates drove the change of biomass in designed treatments within communities.

As shown in Figure 4.4, all the dominant forbs and subordinate forb group significantly increased in biomass from 2018 to 2019 (p=0.005 for *Achillea millefolium,* p=0.000 for *Echinops sphaerocephlus* and p=0.000 for the subordinate group). Biomass of *Echinops sphaerocephalus* had the largest proportional increase (about 367% increase). This appeared to predominantly contribute to the overall increase in forb biomass between the two years.

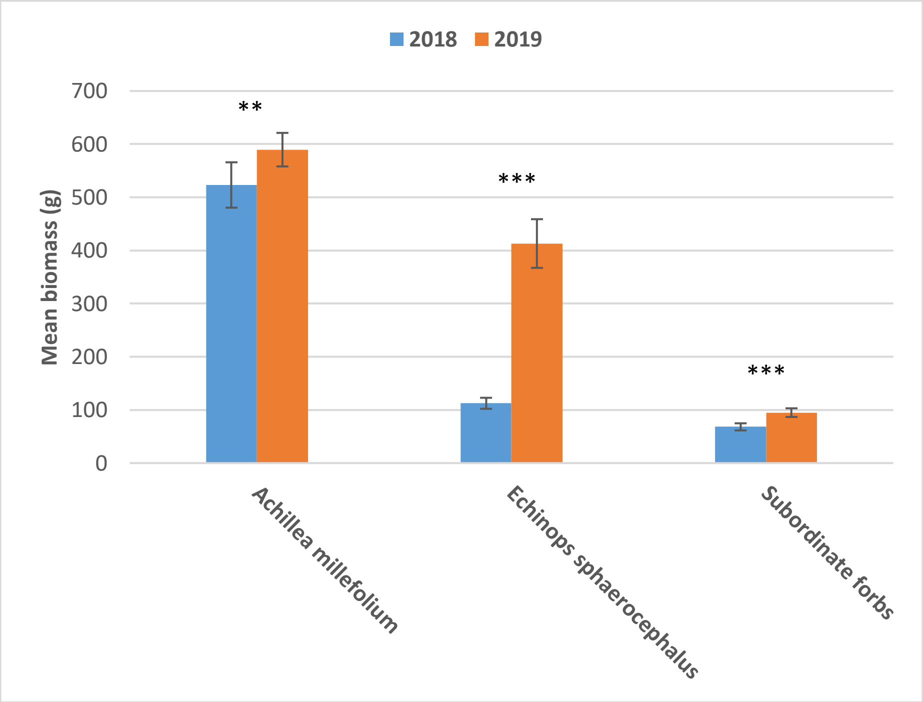


Figure 4.4 Mean dominant forb (*Achillea millefolium* and *Echinops sphaerocephalus*) and subordinate forb (the sum of rest 26 forb species) biomass (g) per plot in August 2018 and August 2019 (*p ≤ 0.05; **p < 0.01, ***p < 0.001 and ns=not significant. Error bar = 2 Standard Errors)

Table 4.2 Effect of time on overall plot results between 2018 and 2019 (SE = Standard Error of Mean; P values refer to the difference between 2018 and 2019)

	2018		**2019**		*P* value
	Mean	SE	Mean	SE	
Forb seedling present number	224.88	16.45	124.61	7.48	0.000 ***
Forb biomass (g)	703.99	50.30	1097.77	47.97	0.000 ***
Grass biomass (g)	303.56	28.68	120.13	12.28	0.000 ***
Forb cover value (%)	39.82	2.68	70.20	2.59	0.000 ***
Grass cover value (%)	34.91	3.38	26.62	2.65	0.000 ***
Bare ground cover value (%)	25.28	2.26	3.29	0.38	0.000 ***
***Achillea millefolium* biomass (g)**	523.27	42.71	589.72	31.52	0.005 **
***Echinops sphaerocephalus* biomass (g)**	112.52	10.34	413.10	45.79	0.000 ***
Subordinate forb biomass (g)	68.19	6.71	94.94	8.16	0.000 ***

4.2.4 Effect of time on forb seedling present number per species

Achillea millefolium seedling numbers were assumed to be the same because they are heavily rhizomatous making it very difficult to establish what an individual *A. millefolium* seedling was, especially at high density. *Platycodon grandiflorum* and *Scutellaria baicalensis* were late emerging species, and hence seedling numbers captured in April across the plots is potentially misleading. Thus, seedling numbers for those three species will not be discussed in this and later chapters.

As shown in Table 4.3, all forb species decreased in terms of number of seedlings from 2018 to 2019 apart from *Geranium pratense* (approximately 6.4 seedlings in 2018 and 6.7 seedlings in 2019 per plot), despite the difference not being statistically significant. Only 3 of subordinate forbs did not significantly decrease numerically. They were *Sanguisorba officinalis* (p=0.604), *Thalictrum aquilegifolium* (p=0.079) and *Veronica longifolia* (p=0.116). These three species that appeared less affected by the passage of time were all tall canopy forbs.

Within short canopy forbs, *Dracocephalum rupestre*, *Dracocephalum ruychiana* and *Thalictrium petaloideum* had the largest seedling numbers at the end of the study, which were approximately 16.5, 11.7 and 11.4. *Potentilla rupestris*, *Galium verum* and *Veronica teucrium* had on average 7.0, 6.9 and 6.1 seedlings per plot at the end of the study.

Origanum vulgare, *Campanula punctata* and *Delphinium grandiflorum* had the largest numbers of seedling within the medium canopy group in both 2018 (27.1, 23.3 and 12.9) and 2019 (15.7, 14.4 and 8.5).

In the tall canopy forb group, *Geranium pratense*, *Thalictrum aquilegifolium* and *Echinops sphaerocephalus* retained the largest seedling numbers in both years which were 6.4, 7.6 and 4.8 in 2018 and 6.7, 6.0 and 3.8 in 2019.

Veronica incana appeared as the least successful species in terms of seedling numbers.

Table 4.3 Effect of time on seedling number of each forb species between 2018 and 2019 (SE = Standard Error of Mean; P values refer to the difference between 2018 and 2019; na = not applicable)

	2018		**2019**		*P* value
	Mean	SE	Mean	SE	
Low canopy					
Shared					
Anemone sylvestris	8.13	0.91	4.96	0.55	0.000 ***
Galium verum	9.30	1.02	6.89	0.65	0.002 **
Potentilla rupestris	9.69	0.81	6.98	0.52	0.000 ***
Pulsatilla vulgaris	7.56	0.90	2.68	0.41	0.000 ***
Veronica teucrium	8.69	0.91	6.08	0.59	0.000 ***
Mongolian					
Dracocephalum rupestre	16.44	1.40	4.85	0.55	0.000 ***

Dracocephalum ruychiana	11.64	1.34	2.42	0.35	0.000 ***
Thalictrum petaloideum	11.40	0.91	5.96	0.53	0.000 ***
Thermopsis lanceolata	8.39	0.95	1.90	0.26	0.000 ***
Veronica incana	0.44	0.16	0.00	0.00	0.004 **
Medium canopy					
Shared					
Achillea millefolium	6.61	0.51	6.61	0.51	na
Origanum vulgare	27.10	2.07	15.67	0.92	0.000 ***
Polemonium caeruleum	7.26	0.66	3.47	0.48	0.000 ***
Stachys officinalis	6.67	0.52	4.14	0.36	0.000 ***
Mongolian					
Campanula punctata	23.25	2.03	14.44	1.32	0.000 ***
Delphinium grandiflorum	12.91	1.26	8.51	0.92	0.000 ***
Kalimeris incisa	10.35	0.97	5.31	0.54	0.000 ***
Platycodon grandiflorum	4.33	0.59	0.03	0.03	na
Scutellaria baicalensis	2.22	0.36	0.19	0.05	na
Tall canopy					
Shared					
Echinops ritro	1.48	0.23	0.78	0.16	0.000 ***
Geranium pratense	6.42	0.46	6.70	0.52	0.079 ns
Sanguisorba officinalis	2.33	0.28	2.00	0.20	0.604 ns
Thalictrum aquilegifolium	7.56	0.67	6.02	0.46	0.079 ns
Veronica longifolia	2.85	0.38	2.34	0.33	0.116 ns
Mongolian					
Aconitum carmichaelii	1.00	0.00	0.83	0.04	0.000 ***
Angelica sylvestris	2.50	0.14	0.63	0.08	0.000 ***
Echinops sphaerocephalus	4.80	0.27	3.81	0.31	0.000 ***
Patrinia scabiosifolia	3.56	0.32	0.43	0.17	0.000 ***

4.2.5 Effect of time on subordinate forb biomass per species

In terms of subordinate forb species, *Thalictrum petaloideum* and *Veronica incana* were considered as the two least successful species. The amount of collected biomass was insufficient to valid the statistical model. The results therefore were not discussed in this thesis.

The subordinate forbs had different responses to the passage of time; 14 species decreased while 10 species increased in biomass from 2018 to 2019 (refer to summarised Table 4.4). Within those species, only 8 species; *Anemone sylvestris, Dephinium grandiflorum, Echinops ritro, Veronica longifolia, Potentilla rupestris, Polemonium caeruleum, Aconitum carmichaelii* and *Angelica sylvetris* had no statistical difference in biomass between the years. Moreover, subordinate forbs that had significant increases in biomass all appeared to be the 'shared' species whereas Inner Mongolian subordinate forbs mostly had significant biomass decreases. Plant canopy height appeared less critical to this pattern.

In both years, most of the subordinate forbs remained tiny and hardly contributed to visual impacts in summer. In general, *Galium verum, Potentilla rupestris, Origanum vulgare, Kalimeris incisa, Geranium pratense, Sanguisorba officinalis* and *Echinops ritro* represented the most abundant subordinate forbs in biomass. Most of the species again appeared as 'shared' species and this pattern was irrespective of the canopy height.

Table 4.4 Biomass change (decrease and increase) of subordinate forb in August 2018 and August 2019 (*p≤0.05; **p≤0.01; ***p≤0.000 and ns=not significant. *Thalictrum petaloideum* and *Veronica incana* had limited data collected and appeared identical in both 2018 and 2019)

Subordinate forb biomass decreased in 2019	**Subordinate forb biomass increased in 2019**
Anemone sylvestre (ns)	*Galium verum* (***)
Pulsatilla vulgaris (***)	*Potentilla rupestris* (ns)
Thermopsis lanceolate (***)	*Veronica teucrium* (*)
Dracocephlum rupestre (***)	*Origanum vulgare* (***)
Dracocephlum ruychiana (***)	*Polemonium caeruleum* (ns)
Campanula punctate (*)	*Stachys officinalis* (**)
Dephinium grandiflorum (ns)	*Geranium pratense* (***)
Kalimeris incisa (***)	*Sanguisorba officinalis* (**)
Platycodon grandiflorum (***)	*Aconitum carmichaelii* (ns)
Scutellaria baicalensis (***)	*Angelica sylvetris* (ns)
Patrinia scabiosifolia (***)	
Echinops ritro (ns)	
Thalictrum aquilegifolium (*)	
Veronica longifolia (ns)	

Table 4.5 Effect of time on biomass of each subordinate forb species between 2018 and 2019 (SE = Standard Error of Mean; P values refer to the difference between 2018 and 2019; na = not applicable)

	2018		**2019**		*P* value
	Mean	SE	Mean	SE	
Low canopy					
Shared					
Anemone sylvestris	0.07	0.02	0.05	0.02	0.075 ns
Galium verum	3.01	0.70	7.64	1.24	0.000 ***
Potentilla rupestris	3.02	0.59	4.20	1.35	0.156 ns
Pulsatilla vulgaris	0.16	0.03	0.04	0.02	0.000 ***
Veronica teucrium	1.32	0.28	1.98	0.41	0.022 *
Mongolian					
Dracocephalum rupestre	2.39	0.29	0.10	0.02	0.000 ***

Dracocephalum ruychiana	0.86	0.13	0.26	0.11	0.000 ***
Thalictrum petaloideum	0.00	0.00	0.00	0.00	0.110 ns
Thermopsis lanceolata	1.18	0.24	0.45	0.13	0.000 ***
Veronica incana	0.00	0.00	0.00	0.00	0.098 ns
Medium canopy					
Shared					
Origanum vulgare	17.99	2.37	42.08	4.45	0.000 ***
Polemonium caeruleum	1.28	0.45	1.22	0.35	0.875 ns
Stachys officinalis	0.40	0.06	1.16	0.27	0.001 **
Mongolian					
Campanula punctata	1.92	0.40	1.04	0.36	0.021 *
Delphinium grandiflorum	2.43	0.52	2.10	0.60	0.364 ns
Kalimeris incisa	18.99	2.72	10.53	2.23	0.000 ***
Platycodon grandiflorum	0.08	0.01	0.01	0.00	0.000 ***
Scutellaria baicalensis	0.61	0.16	0.09	0.03	0.000 ***
Tall canopy					
Shared					
Echinops ritro	3.77	0.88	3.22	0.90	0.499 ns
Geranium pratense	3.57	0.52	12.76	2.16	0.000 ***
Sanguisorba officinalis	2.69	0.55	3.95	0.78	0.002 **
Thalictrum aquilegifolium	0.07	0.01	0.05	0.01	0.024 *
Veronica longifolia	1.51	0.45	1.28	0.31	0.528 ns
Mongolian					
Aconitum carmichaelii	0.26	0.03	0.33	0.06	0.106 ns
Angelica sylvestris	0.03	0.01	0.32	0.26	0.211 ns
Patrinia scabiosifolia	0.57	0.11	0.08	0.02	0.000 ***

4.3 Discussion

4.3.1 The Overall effect of designed treatment on number of forb seedling, forb and grass biomass and cover values

In terms the number of forb seedling and forb biomass, the results broadly agree with previous studies (Del-Val and Crawley, 2005; Dickson and Busby, 2009; Hitchmough, Wagner and Ahmad, 2017) that the number of forbs surviving were much more affected than forb biomass through design. This also suggests a very weak relationship between the number of forb seedlings and biomass production. However, extremely different starting point (i.e. between the ratio of forb: grass of 1: 9 to 9: 1) can enhance the forb biomass in the longer term.

The most of treatments applied significantly affected cover values of forb, grass and bare ground indicating that these can affect community establishment in each spring (April 2018 and 2019). This is useful in terms of community dynamics and appearance as well as the potential weed resistance (Hitchmough, Wagner and Ahmad, 2017). Forb cover was not significantly affected by the depth of sand substrate (p=0.158). This may suggest the forb group was generally insensitive to the levels of moisture stress. There was no significant effect of sowing density on grass cover values (p=0.481) suggesting grasses were likely to emerge and establish earlier in spring (Hitchmough, Kendle and Paraskevopoulou, 2001; Pywell *et al.*, 2003) and increasing sowing density did not increase the coverage due to intra species competition.

The community significantly changed from 2018 to 2019 in all aspects of forb seedling number, forb and grass biomass and cover values. The major shift in community dynamics in the second and third year through for example self-thinning, was also observed in previous studies and the community tended to be more stabilised from the fourth year (Hitchmough and Fleur, 2006).

4.3.2 The effect of time on overall numbers of forb seedling, forb and grass biomass and cover values

The community was forb biomass dominated from 2018, and this pattern was not turned around to become grass dominated as frequently occurred in previous studies (e.g. Bullock et al., 2001, Dickson & Busby, 2009 and Pywell et al., 2003). Forb dominance was facilitated by an increased forb biomass, decreased grass biomass which resulted in a biomass ratio of forb: grass from 2.3: 1 in 2018 to 9.1: 1 in 2019. The establishment of forb biomass suppressed the growth of grass and seemed broadly to be above the threshold to grass competition in this research. Forbs developed far more coverage in spring 2019 than 2018. This suggests that forbs may start to suppress grass by early emergence and imposing light stress from the very beginning of the third growing season. Also, Bjørn et al. (2019) suggested that in designed perennial communities, colonising ability appears less important than the abilities to vegetative survive and persist. This study had an effectively weedy grass seed free substrate that diminished one of the major advantage of grasses; that of high seedling recruitment capacity from the soil seed bank (Edwards and Crawley, 1999). The removal of grass seed heads in June 2018 before the seeds dispersed also decreased the colonisation ability of the *Deschampisa spp.*

The plot management protocol of cutting back all vegetation in June 2017 to improve access of small forbs to light might be as an effective disturbance to restrict grass dominance; as one of the quickest growing species. This mimicked animal grazing processes in a species rich grassland system. Grasses are less palatable to invertebrates which often facilitates their dominance over forbs (Wilby and Brown, 2001), however most forbs used in the research were unpalatable too for minimising this source of experiment bias. It is worth mentioning that one of the most dominant forb species, *Achillea millefolium,* is typically not palatable. The grasses may bounce back in the future as *D. cespitosa* has the character of slow establishment and tend to dominant a community after few years (St. John et al., 2011).

Despite forb biomass increasing, the number of forb seedlings decreased from 2018 to 2019. There were approximately 225 forb seedlings/ plot on average in spring 2018 but only about 125 forb seedlings in 2019. Similar pattern of seedling elimination > than half in the first three growing years was recorded by Hitchmough and Wagner, (2013) and Bjørn *et al.*, (2019). However, individual mass became bigger on average from 3.13g/ plant in 2018 to 8.78g/ plant in 2019. Yoda et al. (1963) discovered the relationship between seedling number and biomass and defined it as “self-thinning”. Accumulating biomass in a community as plants grow causes greater inter and intra specific competition that diminishes seedling number and eventually thins the seedlings down to an optimum number. Significant elimination of seedlings always takes place with the passage of time where the initial density exceeds a critical level. Species diversity also decreased (from 3.55 to 1.72, Shannon Wiener Index) in this study. Loss of species diversity was mainly because of competition from dominants because of a lack of intensive disturbance (Grime, 2002; Sluis, 2002). The loss of seedlings could have been worse had this research not adopted the summer cutback and waited for an autumn cutback. The former gave the subordinates a temporary advantage when the community was subject to a major biomass removal of dominant species (Hitchmough, 2017a). The subordinates had chances to re-emerge to photosynthesise with less light extinction and store more carbohydrates before entering winter dormancy.

4.3.3 The effect of time on biomass of dominant species (*Achillea millefolium* and *Echinops sphaerocephalus*)

A few species often occupy a high proportion of the plant biomass and to control the abundance and fitness of other minor contributors. The extent of dominant species depends upon various factors, for example the availability of resources and the neighbouring species (Grime, 2002). In this research, *Achillea millefolium* and *Echinops sphaerocephalus* were the dominants and occupied 74.33% and 15.98% in 2018; and 53.72% and 37.63% of total forb biomass in 2019.

Achillea millefolium produced far more biomass than other species in both years. It might be the best adapted species to the conditions in this study due to its drought tolerant character, and can be invasive where there is a lack of intensive management (Hurteau, 2003; Bjørn *et al.*, 2019). *Achillea millefolium* is a stress-tolerant competitor (Grime, 2002). It has capacity for massive vegetative production associated with its rhizomatous root system. This character enhances its resource hunting and penetrating abilities by producing more foliage and supressing surrounding species. Rhizomatous species also have better response to disturbance (N'Guessan and Hartnett, 2011). *A. millefolium* had a quicker recovery after cutback in each February and August as well as the cutback in Autumn 2017 which aimed to limit vigorous growth. Besides, *A. millefolium* foliage persists longer than some other species; it is a semi-ever green forb which can supress others from reshoot in spring and autumn.

The mean vegetative production per plots of *Echinops sphaerocephalus* (112.52g) was much less than *A. millefolium* (523.27g) in 2018. *E. sphaerocephalus* was not as competitive to *A. millefolium* as there is a result which suggested a positive coefficient value despite a small correlation (Pearson's correlation efficient = 0.284, p=0.005). However, the biomass climbed by 267% from 112.52g/ plot in 2018 to 413.10g/ plot, the two species started to show a negative relationship as a sign of direct competition in 2019 (Pearson's correlation efficient = -0.283, p=0.005) (the community is shown in Figure 4.5). The rapid growth in 2019 may primarily be due to the taproot system of *E. sphaerocephalus.* This allowed them to access the fertile, moist soil beneath the sand substrate more effective than other species. As one of the biggest species in the community, *E. sphaerocephalus* has potential to produce expansive shoots in terms of the lateral spread and foliage height, even develop a taller layer in the community. Thus, *E. sphaerocephalus* would potentially be more vigorous further enhancing its competitive advantages where more resources were available at the initial stage. Vegetative growth is the most important character to suppress other established plants (Pywell *et al.*, 2003).

E. sphaerocephalus started to emerge only few days after sowing in late May 2017 (as shown in Figure 4.6). Early emergence was helpful for *E. sphaerocephalus* to secure competitive advantages in the seedling stage by developing taller and bigger leaves. The early advantages are perhaps due to the seed size where bigger seeds can increase the survival especially in the dry substrate (Hitchmough, Kendle and Paraskevopoulou, 2001).

However, *E. sphaerocephalus* had a significant loss in seedling number from 2018 to 2019 (about 1.04 seedling loss per plot, p=0.000). Especially, the plots harvested in late August 2018 were observed to have more mortality of *E. sphaerocephalus* than the plots harvested in late July. Late cutback probably disfavoured the big species as they allocated the higher ratio of carbohydrate in shoots rather than roots. Recovery of foliage for photosynthesis to store more chemical energies before dormant is essential to re-emerge in the following year. However, the thinning in seedling number might relieve the intraspecific competition and applied a positive impact on the biomass production of remaining *E. sphaerocephalus.*

Achillea millefolium and *Echinops sphaerocephalus* played the dominant role with different morphological advantages and strategies. *A. millefolium* were probably more tolerant to the nutrient and moisture stress, and effectively responded to cutback, where *E. sphaerocephalus* were able to develop more vigorous shoot and root for a higher level of resource uptake. Both strategies were successful to supress the subordinate species in both years. In order to keep the community species rich for a longer term, early monitoring and intervention, for example removing and thinning the number of individuals, would improve survival of subordinate species.

Unlike *E. sphaerocephalus,* the amount of increase in biomass of *A. millefolium* from 2018 to 2019 was not huge (despite the difference was statistically significant, p=0.005). However, it showed a decrease in the proportion of *A. millefolium* biomass of the total forb biomass. The assumption can be made that in 2018, *A. millefolium* had occupied most possible niches which were ecologically suitable for them. Other species gradually occupied the niches that were not as suitable for *A. millefolium* in 2019. It is reasonable to guess the increase of *A. millefolium* biomass will decrease in future years. As the dominant plant enlarges in population size, intraspecies competition should reduce the vigour of the dominant. One or few competitive or stress-tolerant competitor species within subordinates can replace the dominant (Grime, 2002 and Sluis, 2002). Dominance has high likelihood however to retain for a much longer time period (Silvertown *et al.*, 2006). This study expects that the biomass of *E. sphaerocephalus* and *Deschampsia cespitosa* also persist in the longer term. Climate or change of management scheme may annually reduce their dominance.

Figure 4.5 Grassy dominance was only observed in the plots where the initial grass sowing density was high in summer 2018 (as shown in the picture on the top left corner, taken on the 16th July 2018); however, communities generally became forb dominated in 2019 as *Echinops sphaerocephalus* got much bigger (as shown in the pictures on the top right corner and the bottom, taken on the 29th July 2019).

Figure 4.6. Three big seedlings were *Echinops sphaerocephalus* that were observed as one of the earliest emerging species (picture taken on the 27th May 2017 about 4 days after sowing)

4.3.4 The effect of time on overall number of forb seedlings and biomass of subordinate forb species

A large number of species consistently occupy subordinate position within the hierarchies of herbaceous communities due to ecological and evolutionary process. Their fitness and abundance are heavily controlled by the dominants in the community (Grime, 2002). In a designed dense sown community, this process inevitably takes place as there is not enough physical space and resource to maximise the growth of each individual. Less vigorous species tend to be outcompeted, and this eventually causes a loss of species richness. Enhancing the performance of subordinates, or the small species co-existing with big species, potentially creates a visually flower rich and colourful meadow community, and is politically important to delivery natural aesthetics and the sign of ecological merit in urban contexts to the public (Hitchmough, Paraskevopoulou and Dunnett, 2008; Southon *et al.*, 2017). However, again, this level of richness is not commonly retained in a longer term without effective disturbance or re-establishment of new desirable plants (Bjørn, Weiner and Ørgaard, 2016).

The subordinate group experienced a great diminishment in seedling numbers in this study. Most of the species had significant loss due to competitive elimination mainly from the dominant forbs or grasses (especially at the seedling stage in 2018). The species that persisted best in terms of the number of seedlings (no statistically significant loss in seedling number from 2018 to 2019) were mainly native to the UK (but also in most cases to Inner Mongolia). The nativeness indicates a better adaption to the climate and helps with the fitness. The persistence may also because that the shared species (distributed from Western Europe to Inner Mongolia) tend to have generalist habitat requirements and can grow better than the species have more restricted habitat (Pywell *et al.*, 2003). The better adaption or fitness can help them to survive competition. However, all the low canopy forbs significantly decreased in seedling number irrespective of natural distribution. This predominantly because the selected low canopy forbs are universally shade intolerant (according to Ellengberg indicator values in Table A in Appendices). All the survivor species mostly had taller leaf canopies.

Geranium pratense and *Sanguisorba officinalis* are ones of the few early emerging species that have the characteristics for quick recovery after cutback in both August and February in the community. They also develop taller foliage in the seedling stage. *G. pratense* have long leaf petioles to access the light and shade the surrounding individuals at both seedling and established stages. These advantages also make them more competitive in different stages and produce more biomass. *Veronica longifolia* has strong ability to thrust into the light and were observed specially in the grass dominant plots (refer to Figure 4.7) to do this effectively with its tall leafy stems. Shoot thrust is important to capture light resources to persist especially for those were relatively slow growing (Grime, 2002). Most species persist a similar level of seedling numbers appeared to have upright or elongating stems as an advantage in light competition. On the other hand, *Thalictrum aquilegifolium* had a lower Ellenberg light value of '5' indicating a relatively higher shade tolerance ability within the mix. This is a key strategy to survive in competitive environments (Hitchmough, Paraskevopoulou and Dunnett, 2008; Hitchmough and Wagner, 2013).

The subordinates had different biomass responses to the passage of time. Nativeness appeared more important for the subordinate to the dominant competition. Within the species which had significant biomass difference over the years, 6 out of 6 species that increased biomass were "shared" species whereas 8 of 9 species decreased in biomass were Inner Mongolian species from 2018 to 2019. Small species with rosette form and low basal foliage appeared to be most suppressed over the time (e.g. *Dracocephlum rupestre*). Medium to tall species that suffered decreases of biomass appeared to have late spring emergence of their foliage (e.g. *Platycodon grandiflorum, Scuttellaria baicalensis* and *Patrinia scabiosifolia*). Again, upright or elongating stems in all species appeared to be advantages to produce more standing biomass.

Origanum vulgare, Geranium pratense and *Sanguisorba officinalis* are identified as the key competitive species within the subordinates. All of them are associated with ability of vertical growth. All have quick recovery ability after cutback (Figure 4.8). Besides, *O. vulgare* also have leafy stems and a broad adaptiveness in environmental conditions ('CSR' in competition strategy). This study anticipated that reducing the current dominant species density might lead to *Origanum vulgare, Geranium pratense* or *Sanguisorba officinalis* dominance. *Galium verum* have clambering stems with a higher plasticity which allow them to thrust out from crowded canopies of surrounding species for light hunting. This strategy could significantly increase the biomass production by active light access. Also, this species requires a lower Nitrogen level ('2' in Ellenberg indicator values) which represent the adaptiveness to the sand substrate.

Despite the native species (distributed from Western Europe to Inner Mongolia) appearing to be more persistent, much of this seems due to the key abilities to improve light competition including tall leafy growth form, early emergence and quick recovery after disturbance, allowing both high numbers of seedling and biomass to coexist with dominants. Seedling numbers appeared to be more critical and required the initial light competition strength in the seedling stage. Although *G. verum* and *O. vulgare* significantly increased in biomass, the seedling number diminished. This is likely to be due to the relatively slower growth rate and smaller size in the initial seedling stage. Bigger seedling status can enhance the likelihood to survive. Tremmel and Bazzaz (1993) found that any height advantages gained when plants are small may be compounded over time as they intercept more light and shade their neighbours. In the later chapters, this study will test if any designed treatment can alter the capacity to have greater numbers of forb seedling or biomass of subordinates to reduce the competitive advantages of dominant species.

Figure 4.7 *Veronica longifolia* were often observed to coexist and flowering with dense grass biomass. This might suggest their ability to shoot thrust from grass canopies (picture taken on the 10th July 2019)

Figure 4.8 (from left to right) *Geranium pratense* recovered about a week after cutback in August 2019 (photo taken on 15th Aug 2019); *Sanguisorba officinalis* recovered about 12 days after cutback and already developed taller foliage in August 2019 (photo taken on 20th Aug 2019); *Origanum vulgare* recovered about 12 days after cutback and develop a considerable amount of lateral spread in August 2019 (photo taken on 20th Aug 2019)

Chapter 5 Effects of initial sowing rate

5.1 Introduction

Plant density and diversity are critical for overall community performance and resilience to invasion. This requires an effective process to establish the optimal seedling density and structure. In urban landscapes, the opportunity to start with a “blank sheet” (i.e. relatively weedy species and weedy seed bank free) can reduce the external interference. This advantage may be transient in that the desired species are usually slow growing and hence vulnerable to potential invaders especially weedy grasses. Increased seedling density can be more resistant from invasion however it may diminish the community diversity sooner as asymmetric competition always takes place. Thus, a core question in practice is to what extent can the initial sowing determine the appearance and composition of the established community at least in the short term.

This chapter will investigate that within the same species composition, what are the effects of different sowing densities on forb performance given the potential issues of grass competition and dominance by vigorous species? Does a different starting sowing ratio of forb: grass determine longer term forb establishment?

5.2 Effects of sowing density

Two designed sowing densities refer to target numbers of seedling emergence (forb + grass) of 500/ plot (low sowing density) and 1,000/ plot (high sowing density) regarding estimated emergence % for each species (as explained in Methodology).

5.2.1 Results of sowing density

5.2.1.1 Effect of sowing density on forb seedling numbers, forb and grass biomass and cover values in 2018 and 2019

As shown in Figure 5.1, high sowing density led to significantly more numbers of forb in both April 2018 ($p=0.000$) and 2019 ($p=0.000$). However, doubling sowing density did not double forb seedling number. The difference of forb seedling number between low and high sowing density became smaller in 2019 (ratio of low density: high density=1:1.33) than 2018 (1:1.5).

Within the same treatment, the decrease of forb seedling numbers was significant in both low ($p=0.000$) and high ($p=0.000$) sowing densities. Mortality seemed unavoidable with significant loss of forbs in both treatments. High sowing density (47.4%) had greater mortality than low sowing density (40.6%).

As shown in Figure 5.2, high sowing density did not double the forb biomass (the ratio of low density: high density = 1:1.33) but led to significantly more forb biomass (p=0.015) in August 2018. However, in 2019, there was no significant difference (p=0.710) with forb biomass and this appeared very similar. This differed from the pattern shown for the forb seedling number in 2019. Also, the forb biomass had significant increases in both low and high sowing density (p=0.000 for both).

In terms of grass biomass, Figure 5.3 showed that low sowing density had higher but not significantly higher grass biomass than high sowing density in 2018 (p=0.443, the ratio of low density: high density = 1: 0.89). In 2019, the proportional difference (low density: high density = 1: 0.60) of grass in two sowing densities became bigger despite the total amount of biomass decreasing (p=0.000 for both sowing densities). Grass biomass became significantly (p=0.002) greater in low sowing density than high sowing density in 2019.

Figure 5.4 – 5.6 show that high sowing density led to a significantly higher forb (p=0.000) cover value and significantly decreased bare ground coverage (p=0.000) than the low sowing density in 2018. However, increasing sowing density did not increase grass coverage in April 2018 (p=0.960). In 2019, despite forb coverage appearing higher in the high sowing density grass coverage being higher in the low sowing density treatment, these treatments had no significant effect on the cover values.

i) Forb seedling numbers present per plot

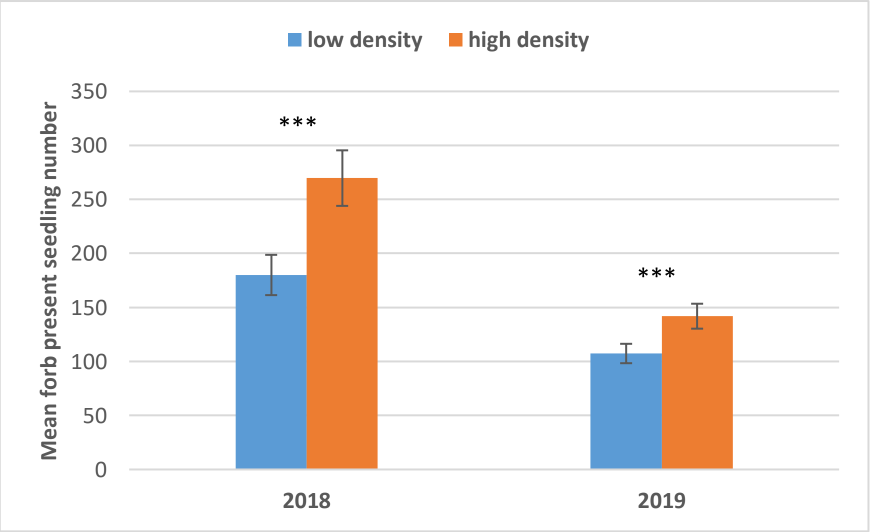


Figure 5.1 Effect of sowing density (low and high density) on forb seedling number/ plot in April 2018 and April 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

ii) Forb and grass biomass per plot

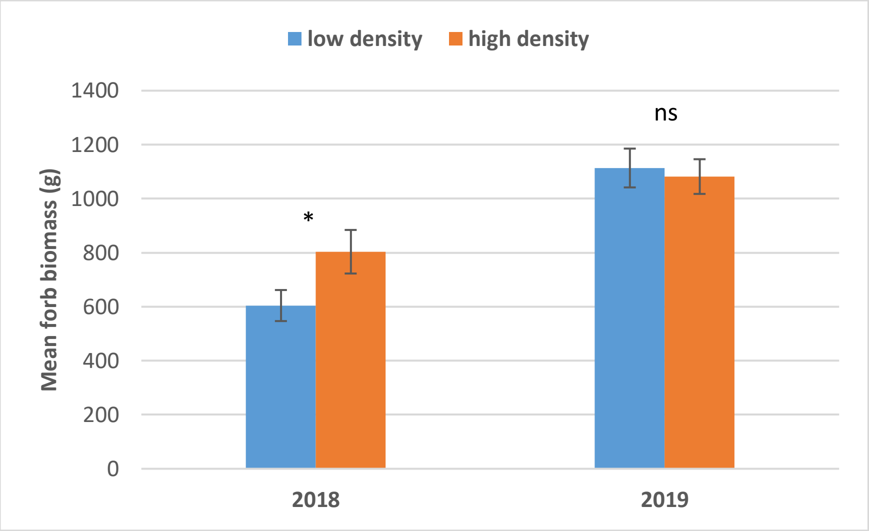


Figure 5.2 Effect of sowing density (low and high density) on forb biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.001 and ns=not significant. Error bar = 2 Standard Errors)

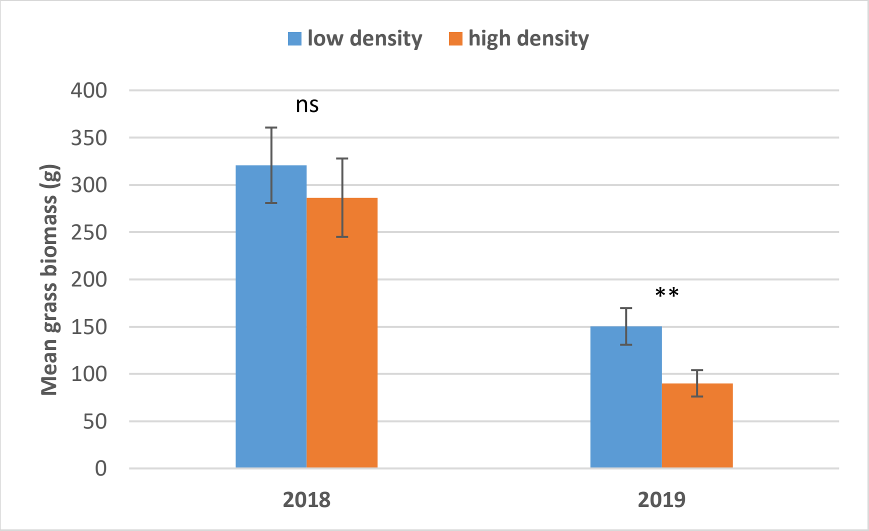


Figure 5.3 Effect of sowing density (low and high density) on grass biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

iii) Forb, grass and bare ground cover values

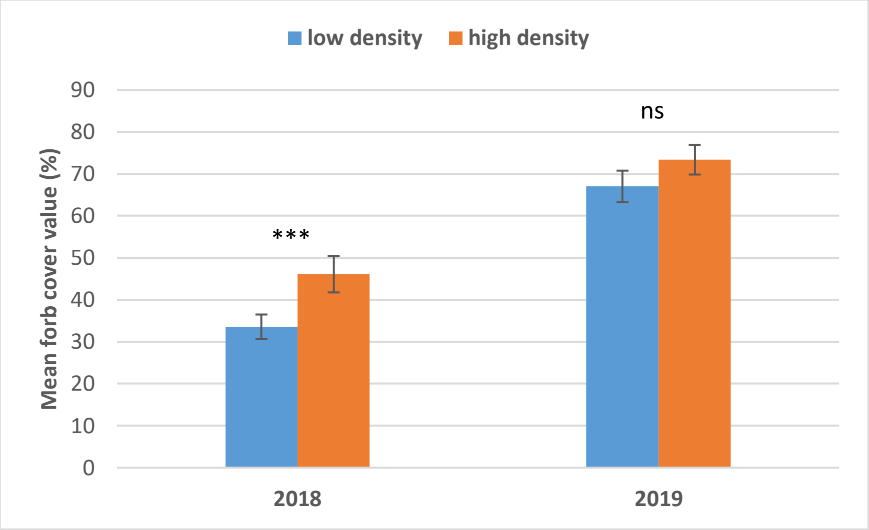


Figure 5.4 Effect of sowing density (low and high density) on forb cover values in May 2018 and May 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

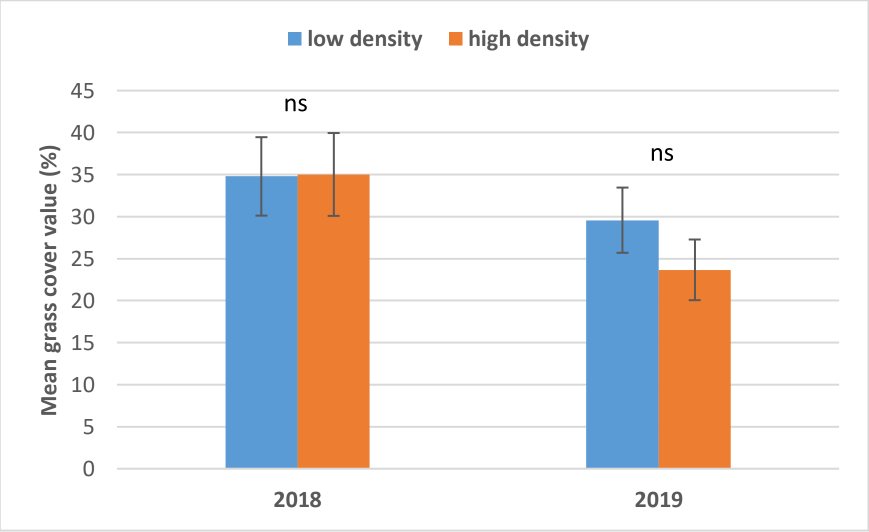


Figure 5.5 Effect of sowing density (low and high density) on grass cover values in May 2018 and May 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

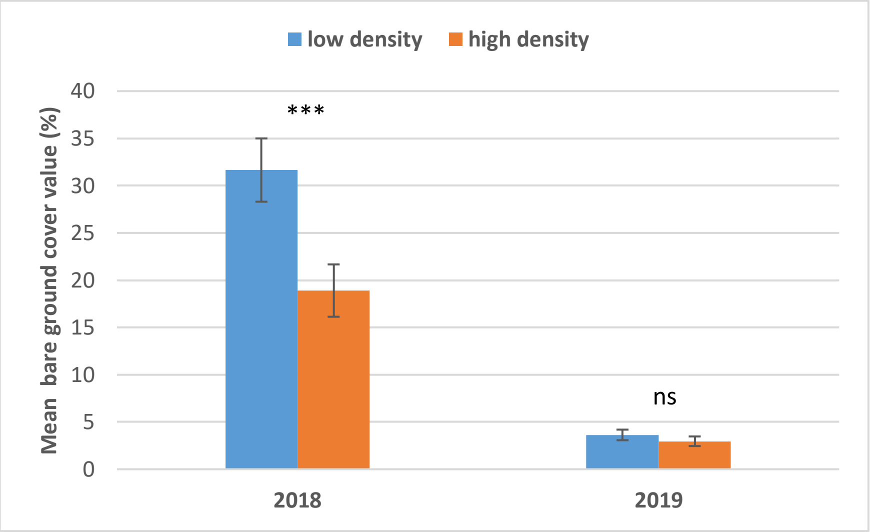


Figure 5.6 Effect of sowing density (low and high density) on bare ground cover values in May 2018 and May 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

5.2.1.2 Effect of sowing density on dominant forbs (*Achillea millefolium* and *Echinops sphaerocephalus*) and subordinate forb biomass in 2018 and 2019

High sowing density led to significantly more biomass of *Achillea millefolium* in August 2018 (p=0.004) but became less effective to increase *A. millefolium* biomass in 2019 (p=0.056, marginal significance level) (Figure 5.7). However, *Echinops sphaerocephalus* biomass had no significant difference between the two treatments in both years (p=0.997 in 2018 and p=0.171 in 2019) despite low sowing density led to more *E. sphaerocephalus* biomass (467.44g) than the high sowing density treatment (358.76g) in 2019 (Figure 5.8). Subordinate biomass showed a similar pattern to *E. sphaerocephalus* (p=0.622 in 2018 and p=0.139 in 2019) and had more biomass in the low sowing density treatment (104.88g comparing with 85.01g in the high sowing density) despite not being significant (Figure 5.9).

High sowing density led to significantly more forb biomass in 2018, this was mostly because of the higher biomass productivity of *Achillea millefolium* in high sowing density which predominantly represented forb biomass in 2018.

Looking into the biomass change within the same treatment between the two years, *E. sphaerocephalus* (p=0.000 for both) and the subordinates (p=0.000 in low density and p=0.019 in high density) had significantly increased biomass in both sowing density treatments. However, *A. millefolium* biomass significantly increased in low density (p=0.000) but no significant difference in high density from 2018 to 2019 (p=0.635). This suggests the high sowing density led *A. millefolium* to reach the maximise biomass productivity sooner.

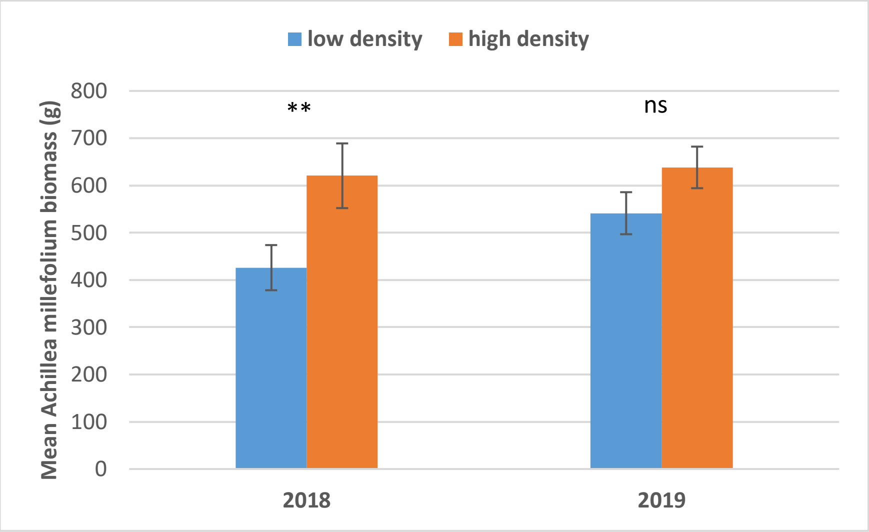


Figure 5.7 Effect of sowing density (low and high density) on *Achillea millefolium* biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

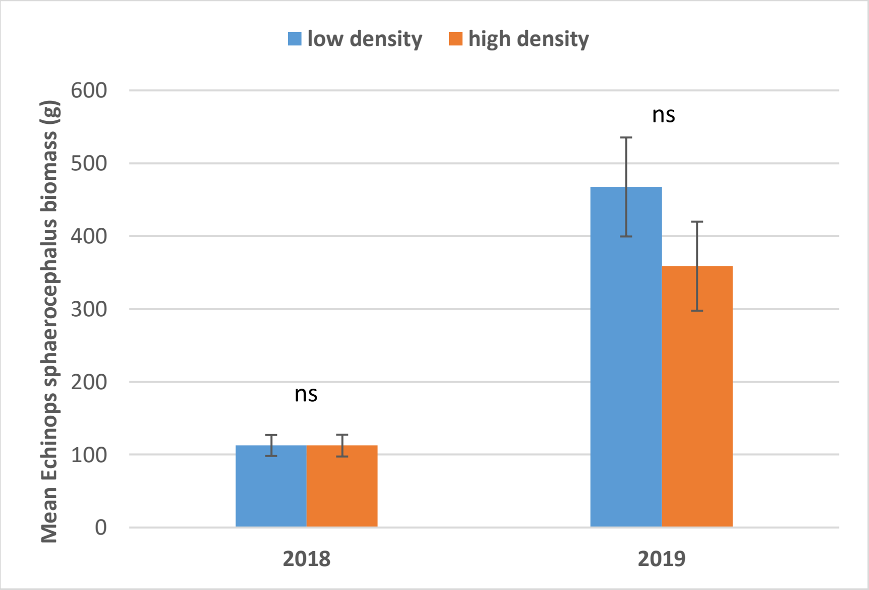


Figure 5.8 Effect of sowing density (low and high density) on *Echinops sphaerocephalus* biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

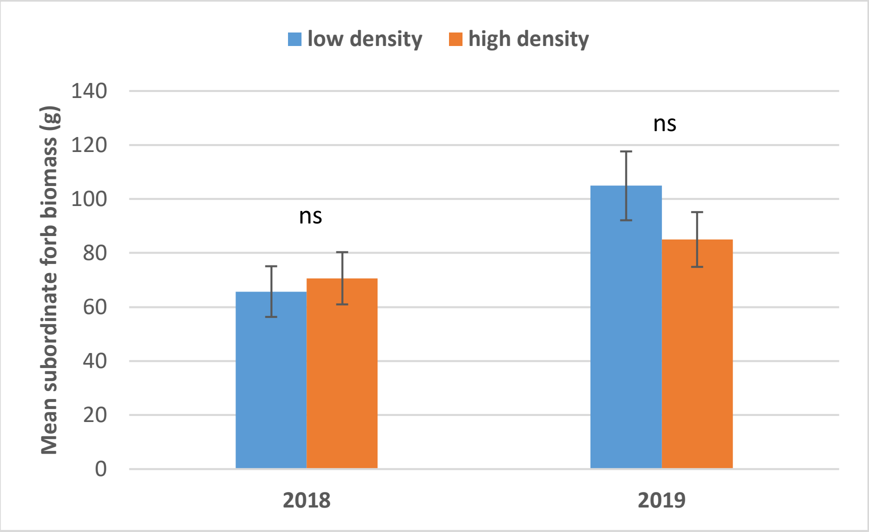


Figure 5.9 Effect of sowing density (low and high density) on subordinate forb biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

Table 5.1 Effect of sowing density on overall plot results in 2018 and 2019 (SE = Standard Error of Mean)

	2018					**2019**				
	Low density		**High density**		*P* value	**Low density**		**High density**		*P* value
	Mean	SE	Mean	SE		Mean	SE	Mean	SE	
Forb seedling present number	180.04	18.64	269.71	25.71	0.000 ***	107.35	8.98	141.88	11.54	0.000 ***
Forb biomass (g)	604.34	57.46	803.63	80.64	0.015 *	1113.59	71.84	1081.95	64.27	0.710 ns
Grass biomass (g)	320.70	39.91	286.43	41.47	0.401 ns	150.21	19.39	90.04	13.96	0.002 **
Forb cover value (%)	33.56	2.94	46.08	4.32	0.000 ***	67.02	3.75	73.38	3.54	0.098 ns
Grass cover value (%)	34.79	4.67	35.02	4.93	0.960 ns	29.58	3.88	23.67	3.61	0.139 ns
Bare ground cover value (%)	31.65	3.35	18.90	2.77	0.000 ***	3.63	0.56	2.96	0.51	0.331 ns
***Achillea millefolium* biomass (g)**	426.06	47.77	620.49	68.48	0.004 **	541.27	44.50	638.18	44.00	0.056 ns
***Echinops sphaerocephalus* biomass (g)**	112.56	14.39	112.48	15.00	0.997 ns	467.44	67.95	358.76	61.11	0.171 ns
Subordinate forb biomass (g)	65.73	9.38	70.66	9.67	0.622 ns	104.88	12.75	85.01	10.14	0.139 ns

5.2.1.3 Effect of sowing density on forb seedling number per species in 2018 and 2019

Table 5.2 showed that in 2018, 18 out of 23 subordinate forb species (collected data for *Platycodon gradiflorum*, *Scutellaria baicalensis* and *Aconitum carmichaelii* not validated for the statistical model) had significantly higher seedling numbers in high sowing density treatments. However, in 2019, the species number went down in 9 out of 24 (collected data for *Platycodon gradiflorum* and *Scutellaria baicalensis* not validated for the statistical model). The effect of increasing sowing density to increase numbers of seedlings was reduced with the passage of time in most of the species especially the lower canopy forb species. Eight low canopy subordinates had more seedlings in high density in 2018 and only

2 species retained this advantage in 2019 (*Potentilla rupestris*, p=0.001 and *Veronica teucrium*, p=0.021). However, 5 medium and 6 tall canopy forbs had significantly more seedlings and the number reduced to 4 and 4.

Overall, seven forb species had significantly higher numbers of seedling in high sowing density in both 2018 and 2019. The species were *Potentilla rupestris* (p=0.003 in 2018, p=0.001 in 2019), *Origanum vulgare* (p=0.000, p=0.007) *Polemonium caeruleum* (p=0.000, p=0.003), *Kalimeris incisa* (p=0.000, p=0.000), *Stachys officinalis* (p=0.000, p=0.000), *Thalictrum aquilegifolium* (p=0.001, p=0.000) and *Echinops sphaerocephalus* (p=0.000, p=0.000).

Table 5.2 Effect of sowing density on seedling number of each forb species in 2018 and 2019 (SE = Standard Error of Mean; na = not applicable)

	2018					**2019**				
	Low density		**High density**		*P* value	**Low density**		**High density**		*P* value
	Mean	SE	Mean	SE		Mean	SE	Mean	SE	
Low canopy										
Shared										
Anemone sylvestris	6.63	1.17	9.63	1.36	0.003 **	4.92	0.88	5.00	0.67	0.325 ns
Galium verum	7.44	1.15	11.17	1.64	0.012 *	6.02	0.82	7.75	1.01	0.061 ns
Potentilla rupestris	8.19	1.02	11.19	1.22	0.003 **	5.81	0.60	8.15	0.82	0.001 **
Pulsatilla vulgaris	5.27	0.86	9.85	1.53	0.001 **	2.46	0.47	2.90	0.68	0.363 ns
Veronica teucrium	7.73	1.17	9.65	1.40	0.134 ns	5.33	0.59	6.83	1.02	0.492 ns
Mongolian										
Dracocephalum rupestre	13.21	1.58	19.67	2.24	0.000 ***	4.96	0.80	4.75	0.77	0.393 ns
Dracocephalum ruychiana	9.83	1.54	13.44	2.18	0.017 *	2.42	0.53	2.42	0.47	0.362 ns
Thalictrum petaloideum	8.75	1.08	14.04	1.38	0.000 ***	4.98	0.59	6.94	0.86	0.021 *
Thermopsis lanceolata	5.98	0.89	10.79	1.60	0.000 ***	1.60	0.32	2.19	0.41	0.059 ns
Veronica incana	0.33	0.14	0.54	0.29	0.984 ns	0.00	0.00	0.00	0.00	0.844 ns
Medium canopy										
Shared										
Achillea millefolium	4.27	0.41	8.96	0.81	0.000 ***	4.27	0.41	8.96	0.81	0.000 ***
Origanum vulgare	21.79	2.15	32.42	3.38	0.000 ***	13.60	1.15	17.73	1.38	0.007 **
Polemonium caeruleum	5.54	0.72	8.98	1.05	0.000 ***	2.71	0.45	4.23	0.84	0.003 **
Stachys officinalis	4.33	0.42	9.00	0.83	0.000 ***	2.90	0.34	5.38	0.60	0.000 ***
Mongolian										
Campanula punctata	21.85	2.78	24.65	2.98	0.128 ns	14.27	1.85	14.60	1.90	0.802 ns
Delphinium grandiflorum	11.15	1.58	14.67	1.94	0.002 **	7.67	1.12	9.35	1.46	0.082 ns
Kalimeris incisa	6.52	0.81	14.19	1.58	0.000 ***	3.90	0.53	6.73	0.90	0.000 ***
Platycodon grandiflorum	3.33	0.70	5.33	0.93	na	0.00	0.00	0.06	0.06	na
Scutellaria baicalensis	2.15	0.48	2.29	0.56	na	0.23	0.09	0.15	0.06	na
Tall canopy										
Shared										
Echinops ritro	1.04	0.20	1.92	0.40	0.060 ns	0.48	0.13	1.08	0.28	0.051 ns
Geranium pratense	5.17	0.51	7.67	0.73	0.004 **	5.56	0.61	7.83	0.82	0.039 *
Sanguisorba officinalis	1.52	0.23	3.15	0.49	0.000 ***	1.60	0.23	2.40	0.33	0.040 *
Thalictrum aquilegifolium	5.92	0.68	9.21	1.12	0.001 **	4.67	0.47	7.38	0.75	0.000 ***
Veronica longifolia	2.44	0.46	3.27	0.60	0.210 ns	2.23	0.43	2.46	0.50	0.830 ns
Mongolian										
Aconitum carmichaelii	1.00	0.00	1.00	0.00	na	0.83	0.05	0.83	0.05	0.980 ns
Angelica sylvestris	2.00	0.12	3.00	0.24	0.000 ***	0.75	0.12	0.50	0.09	0.119 ns
Echinops sphaerocephalus	3.75	0.27	5.85	0.41	0.000 ***	2.85	0.31	4.77	0.49	0.000 ***
Patrinia scabiosifolia	2.92	0.36	4.21	0.51	0.001 **	0.33	0.10	0.52	0.32	0.271 ns

5.2.1.4 Effect of sowing density on subordinate forb biomass per species in 2018 and 2019

Regarding Table 5.3, increasing sowing density generally had limited capacity to increase subordinate forb biomass for most of the species in both years. *Stachys officinalis* was the only subordinate forb that had significantly more biomass in high sowing density treatment in 2018 (p=0.002). Low sowing density however increased the biomass of *Patrinia scabiosifolia* (p=0.032) in 2018, *Origanum vulgare* (p=0.006) in 2019 and *Dracocephalum rupestre* in both years (p=0.018 in 2018 and p=0.005 in 2019). Despite no statistical difference (p=0.106), *Potentilla rupestris* produced far more biomass in low sowing density (6.17g) comparing with high sowing density (2.23g).

Table 5.3 Effect of sowing density on biomass of each subordinate forb species in 2018 and 2019 (SE = Standard Error of Mean)

	2018					**2019**				
	Low density		**High density**		*P* value	**Low density**		**High density**		*P* value
	Mean	SE	Mean	SE		Mean	SE	Mean	SE	
Low canopy										
Shared										
Anemone sylvestris	0.08	0.02	0.06	0.02	0.536 ns	0.06	0.04	0.04	0.02	0.481 ns
Galium verum	2.34	0.73	3.69	1.20	0.243 ns	6.76	1.34	8.52	2.09	0.391 ns
Potentilla rupestris	3.70	1.09	2.35	0.46	0.190 ns	6.17	2.61	2.23	0.58	0.106 ns
Pulsatilla vulgaris	0.18	0.06	0.13	0.04	0.429 ns	0.03	0.02	0.05	0.03	0.672 ns
Veronica teucrium	1.21	0.27	1.42	0.49	0.664 ns	2.32	0.66	1.64	0.48	0.305 ns
Mongolian										
Dracocephalum rupestre	2.94	0.49	1.84	0.31	0.018 *	0.16	0.05	0.03	0.01	0.005 **
Dracocephalum ruychiana	1.07	0.23	0.66	0.13	0.066 ns	0.43	0.22	0.09	0.05	0.091 ns
Thalictrum petaloideum	0.00	0.00	0.00	0.00	na	0.00	0.00	0.00	0.00	0.140 ns
Thermopsis lanceolata	1.13	0.40	1.22	0.27	0.839 ns	0.30	0.14	0.60	0.22	0.214 ns
Veronica incana	0.00	0.00	0.00	0.00	0.400 ns	0.00	0.00	0.00	0.00	1.000 ns
Medium canopy										
Shared										
Origanum vulgare	20.15	3.59	15.84	3.10	0.257 ns	52.53	7.19	31.63	4.89	0.006 **
Polemonium caeruleum	0.56	0.26	2.01	0.86	0.056 ns	0.77	0.24	1.66	0.65	0.142 ns
Stachys officinalis	0.26	0.05	0.54	0.10	0.002 **	0.76	0.16	1.57	0.51	0.102 ns
Mongolian										
Campanula punctata	2.21	0.58	1.64	0.56	0.415 ns	1.24	0.39	0.84	0.61	0.545 ns
Delphinium grandiflorum	2.26	0.62	2.59	0.85	0.713 ns	2.01	0.73	2.18	0.97	0.875 ns
Kalimeris incisa	15.66	3.31	22.33	4.31	0.111 ns	10.85	3.34	10.21	2.98	0.866 ns
Platycodon grandiflorum	0.10	0.03	0.07	0.01	0.315 ns	0.02	0.01	0.00	0.00	0.068 ns
Scutellaria baicalensis	0.73	0.26	0.50	0.17	0.392 ns	0.10	0.04	0.08	0.05	0.609 ns
Tall canopy										
Shared										
Echinops ritro	2.71	0.85	4.84	1.54	0.199 ns	2.73	1.05	3.71	1.47	0.564 ns
Geranium pratense	3.66	0.63	3.48	0.82	0.847 ns	11.87	2.14	13.65	3.77	0.636 ns
Sanguisorba officinalis	2.48	0.62	2.90	0.92	0.678 ns	3.55	0.96	4.36	1.25	0.575 ns
Thalictrum aquilegifolium	0.08	0.02	0.07	0.02	0.674 ns	0.07	0.03	0.02	0.01	0.077 ns
Veronica longifolia	1.14	0.46	1.87	0.77	0.369 ns	1.53	0.48	1.04	0.41	0.402 ns
Mongolian										
Aconitum carmichaelii	0.29	0.05	0.22	0.03	0.231 ns	0.42	0.10	0.24	0.05	0.087 ns
Angelica sylvestris	0.02	0.01	0.03	0.02	0.741 ns	0.08	0.03	0.55	0.52	0.329 ns
Patrinia scabiosifolia	0.78	0.20	0.37	0.10	0.032 *	0.10	0.04	0.06	0.02	0.363 ns

5.2.2 Discussion on effects of sowing density

5.2.2.1 The effect of sowing density on overall forb seedling number, forb and grass biomass and cover values

High sowing density increased the number of forb seedlings present but not the forb biomass in the long term. This agrees with the previous study of Dickson and Busby (2009). The general pattern in relation to seedling numbers, was for this to initially increase with the greater number of seeds sown in the high density treatment (p=0.000 in 2018) but for this then to decline with time, presumably because of greater competition and competitive elimination within the high density sowings. Forb seedling numbers were however still higher in the third year (p=0.000). This suggests that putting more forb seeds in the mix can produce more forb individuals at least for three years in a design project, but that beyond this time period forb seedling numbers are likely to become the same at both sowing densities, as this is determined by other factors such as competition between individuals and species, and also potentially by herbivory. In some cases, having more seedlings present for up to three years may be useful, as it may lead to more attractive appearance sooner and may reduce invasion of weeds due to greater competition.

Seedling number did not linearly correspond to the sowing density. Doubling sowing density only increased the number of forb individuals by about 90 (180 in low sowing density and 270 in high sowing density) in 2018 and it shrunk to about 35 (107 in low sowing density and 142 in high sowing density) in 2019. This differs from the results of Hitchmough et al., (2008) in which doubling sowing density almost achieved twice the number of forb seedlings in all three experiment years. The most likely explanation for this is that in this experiment the sowing densities and emergences were higher, and or the seedlings used were more sensitive to shade, and hence competitive elimination was more marked. In term of increasing the biomass for forb species, the high forb sowing density was effective in 2018 (p=0.015) but had no effect on forb biomass in 2019. The result indicates that doubling sowing density can speed up the time taken to reach a forb biomass ceiling faster but cease when the biomass increases in the third year. This also differs from Hitchmough et al., (2008) but agrees with the findings of Lubin *et al.*, (2019) who found that high sowing rate of component species is unlikely to make a grassland community more abundant in forb biomass in the long term.

High sowing density initially made more biomass and created a higher self-thinning rate (Westoby, 1981). The main source of competitive stress in the research of Hitchmough et al., (2008) was from spontaneous ruderal species on site. Seedling loss in this research was mainly due to the inter or intra species competition within the designed mix. Doubling sowing density may however be effective in landscape projects where the topsoil has a large weed seed bank (Stevenson, Bullock and Ward, 1995; Wagner, Walker and Pywell, 2018). However, a low sowing density appears to be sufficient to establish an abundant meadow community on a well-prepared weed free site (such as with sand mulches) with a moderate level of disturbance (Lubin *et al.*, 2019; Scotton, 2019). It is not possible in the longer term to override ecological processes such as self-thinning, by adding more and more seed. Of course, if too low a seed density is sown, this may lead to no or little impact, so a balance has to be struck. In the author’s study, species diversity was similar within the treatments in both years (Shannon Wiener index: 3.04 in low density 2018; 3.08 in high density 2018; 2.94 in low density 2019 and 2.97 in high density 2019).

Forb cover values measured in May 2018 and 2019 reflect the pattern of forb biomass in August 2018 and 2019. Cover values in Spring seem to be reliable predictors of biomass in Summer. The capacity of high sowing density to achieve significantly higher vegetative cover value (p=0.000) in April 2018 and prevent potential weeds invasion and deliver evidence of ‘ecological value’ may be politically important (Hitchmough, Paraskevopoulou and Dunnett, 2008).

Grasses produced more biomass in the low forb density treatment in both 2018 and 2019 (not significant in 2018, p=0.443 but significant in 2019, p=0.002). At the outset of the study the grass *Deschampsia cespitosa* had been seen as being the likely dominant in the experiment but surprisingly it turned out to be less competitive than the forbs. Where forb biomass was lower, it allowed grass to be more persistent in both coverage in spring and biomass in summer. *Deschampsia cespitosa* was outcompeted by forbs rather than being subject to self-thinning by other invading grasses, as these were largely absent from the communities.

5.2.2.2 The effect of sowing density on biomass of the dominants (*Achillea millefolium* and *Echinops sphaerocephalus*) and the subordinate species

High sowing density only increased the biomass of *Achillea millefolium* in 2018. This suggests that sowing more seeds of every species in the mix may favour the most competitive species, and supress most of the rest in the community, at least in a short term. Subordinates which are relatively competitive (to be shown in the next section) can increase their biomass but the increase did not alter the overall community structure. In 2019, *Echinops sphaerocephalus* and subordinates produced more biomass in the low as opposed to high sowing density although the difference was not statistically significant. This is presumably because *A. millefolium* were relatively less abundant in this treatment and left more space to allow more biomass production by other species. Dickson and Busby (2009) found that spatial separation is an effective way to encourage the growth of less competitive species in a community. However, in practice lower sowing rates may be less effective in soils with high soil weed seed banks.

As previously mentioned, grass biomass declined more in the high rather than low density treatment. This is mostly likely due to the greater presence of *A. millefolium* in the treatment. Dwyer (1958) found that the greater the biomass of rhizomatous forb, like *A. millefolium,* the greater the amount of reduction in grass biomass.

5.2.2.3 The effect of sowing density on subordinate forb species

It is an attractive idea that by increasing the number of sown seed of species that are likely to be subordinate, they will be more abundant and have a bigger biomass. However, the results suggest that increasing sowing density can increase the number of subordinate seedlings but unlikely to enhance the subordinate biomass. Abilities to access light resources through early emergence and projecting taller leafy foliage are essential to survive the more intense competition from dominants. Also, species that have wider geographical distributions tend to be more likely to survive. However, as the dominant species established, subordinates were generally suppressed with similar biomass across the species in both treatments. Potential advantages in light competition barely benefitted any species to produce more biomass in the higher sowing density. This suggests that the dominant competition is highly asymmetric compared with the competitiveness of subordinate species.

Despite that the competition of grasses and the dominant forb *Achillea millefolium* became greater with the higher sowing density, this treatment was effective in increasing the seedling number of most of subordinate forb species (18 out of 23) in 2018. Although a few species such as *Veronica teucrium* or *Campanula punctata* did not show a statistically significant difference, seedling numbers were still higher in high sowing density. The effect of sowing density on subordinate seedling number was reduced in 2019. Only 9 species still had significantly more seedling numbers in the high density treatments comparing. Seven species, out of these 9, were native to the UK suggesting that species with native distributions are potentially more adaptive to stress and more persistent in UK climate. Also, most of them showed advantages of light competitiveness from the seedling stage. Those advantages include early emergence and growth (i.e. *Potentilla rupestris* and *Polemonium caeruleum*), rapid seedling growth (i.e *Origanum vulgare*) or advantages in architecture like long petioles and tall leafy stems (i.e. *Geranium pratense, Kalimeris incisa* and *Sanguisorba officinalis*) to remain more seedling numbers in high sowing density.

However, the high sowing density treatment barely led to a significant difference on subordinate species biomass from 2018. The biomass of subordinate forbs responded to the higher sowing density in different ways. In 2018, 11 forb species increased their biomass in the high sowing density while 13 species decreased (*Thalictrum petaloideum* and *Veronica incana* were particularly poorly represented). Although the species which have tall leafy, elongating or clambering stems such as *Galium verum, Geranium pratense and Sanguisorba officinalis* produced more biomass in the high sowing density, the effect was not significant.

Stachys officinalis was the only species that had a significant increase of biomass among the subordinates. Due to seed dormancy problems, *Stachys officinalis* was one of the species established in the experiment by planting, so each plant had greater size when they were plugged in than many of other subordinates in the plots, and this helped them to cope with shading stress generated by other dominant species. Also, the biomass increased linearly within the high density treatment. Transplantation can be a possible way to give species, that are difficult to germinate, a better starting point. This can help to achieve a desired density in association with a designed mix, at the beginning. This pattern in *Stachys officinalis* appeared again in the ‘sowing ratio of forb: grass’ treatment. However, *Angelica sylvestris,* also one of the transplanted species remained very small size (0.02 and 0.03g in low and high sowing density treatments). Lack of ecological adaptiveness to the community presumably caused this, this species is largely associated with wet sites, and it was probably too dry for this species to establish in this site (the highest rating of ‘8’ in the Ellenberg’s value for moisture within this forb group).

Dracocephlum rupestre and *Patrinia scabiosifolia* showed a significantly negative response to the doubling of sowing density. Where the competitive stress was lower, they established more biomass in 2018. *D. rupestre* has low basal foliage and low comparative biomass production rate placing it at a disadvantage in light competition where the vegetative coverage was high. Despite the statistically insignificant difference, species such as *Potentilla rupestris* or *Campanula punctata* which have low foliage also showed the negative response to doubling the sowing density. *Patrinia scabiosifolia* is also a late emerging species, meaning it has to compete for light with already actively growing species.

In 2019, (despite there still being no significance, no species had higher mean biomass in the high sowing density, suggesting suppression by *Achillea millefolium* and *Echinops sphaerocephalus*) *Origanum vulgare* produced a biomass of 52.53g in low density and 31.63 in high density. As the most abundant subordinate species *O. vulgare* seemed to take the advantage of *Achillea millefolium* at low sowing density and produced significantly more biomass. *Origanum vulgare* emerged early and the green foliage lasts longer, and these are probably the main characters making it competitive.

5.3 Effects of sowing ratio of forb: grass

The sowing ratios of forb: grass was designed according to estimates on target numbers of seedling emergence on forbs and grasses. The treatments will be referred as 10% forb (sowing ratio of forb: grass = 1: 9), 50% forb (sowing ratio of forb: grass = 1: 1) and 90% forb (sowing ratio of forb: grass = 9: 1) in this and later chapters.

5.3.1 Results of sowing ratio of forb: grass

5.3.1.1 Effect of sowing ratio of forb: grass on forb seedling numbers, forb and grass biomass and cover values in 2018 and 2019

Figure 5.10 shows that increasing sowing ratio of forb: grass significantly increased forb seedling number in 2018 and 2019 (p=0.000 for all). However, this did not lead to a linear increase. The number of forb seedlings diminished with increasing the ratio of forb: grass. The forb sowing ratio treatments of 10%: 50%: 90% forb across all three treatments led to an established forb seedling ratio of approximately 1: 3.4: 5.0 in 2018 and 1: 2.3: 3.4 in 2019. This suggested higher competitive mortalities in higher forb: grass ratios.

The higher sowing ratios of forb: grass was generally less effective in increasing forb biomass production (as shown in Figure 5.11). In 2018, the 50% forb and 90% forb treatment led to significantly more forb biomass than the 10% forb treatment (p=0.000 for both) but there was no significant difference in forb biomass between the 50% and 90% treatment (p=0.382). In 2019, the effect was further reduced and only increasing the sowing ratio of forb: grass from 1:9 to 9: 1 significantly increased forb biomass (p=0.006). Similar to the sowing density treatment, increasing forb sowing density only led forb biomass to achieve the ceiling faster but did not increase forb biomass in the longer term.

In terms of grass biomass, higher sowing ratios of grass: forb increased grass biomass in 2018 although the trend was not linear (p=0.002 between the 10% grass and 50% grass, p=0.034 between the 50% grass and 90% grass and p=0.000 between the 10% and 90% treatment) (as shown in Figure 5.12). A similar pattern was shown in 2019; only the 10% forb treatment had significantly more grass biomass than the 50% and 90% (p=0.000 for both) but no significant difference between the 50% and 90% forb treatment (p=0.340).

In spring 2018, forb coverage increased with increasing sowing ratio of forb: grass (p=0.000 between the 10% and 50%, and the 10% and 90% treatment, p=0.007 between the 50% and 90% treatment) despite the increase not being linear (Figure 5.13). Despite the difference became smaller in 2019 especially between the 50% and 90% treatment (p=0.030), increasing the ratio of forb: grass still significantly increase forb cover values in spring. Grass cover values showed a similar pattern to grass biomass that the higher ratio of grass to forb significantly increased grass cover values in 2018 (Figure 5.14). However, there was no significant difference between the 50% and 90% treatment in 2019 (p=0.063). In terms of bare ground cover values, no treatment made significant effect. Changing the sowing ratio of forb: grass did not affect the overall amount of vegetative coverage (as shown in Figure 5.15).

i) Forb seedling numbers present per plot

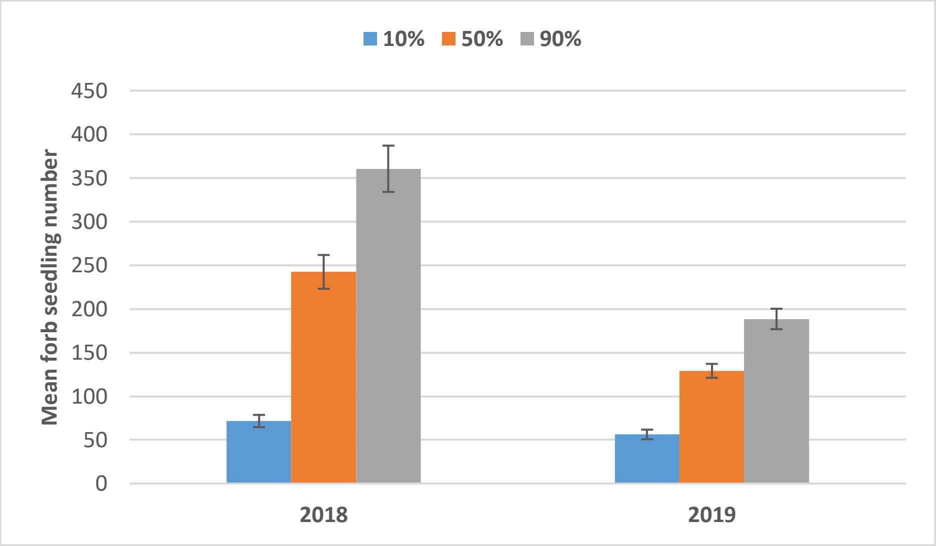


Figure 5.10 Effect of sowing ratio of forb: grass on forb present seedling number/ plot in April 2018 and April 2019 (bar = 2 Standard Errors)

ii) Forb and grass biomass per plot

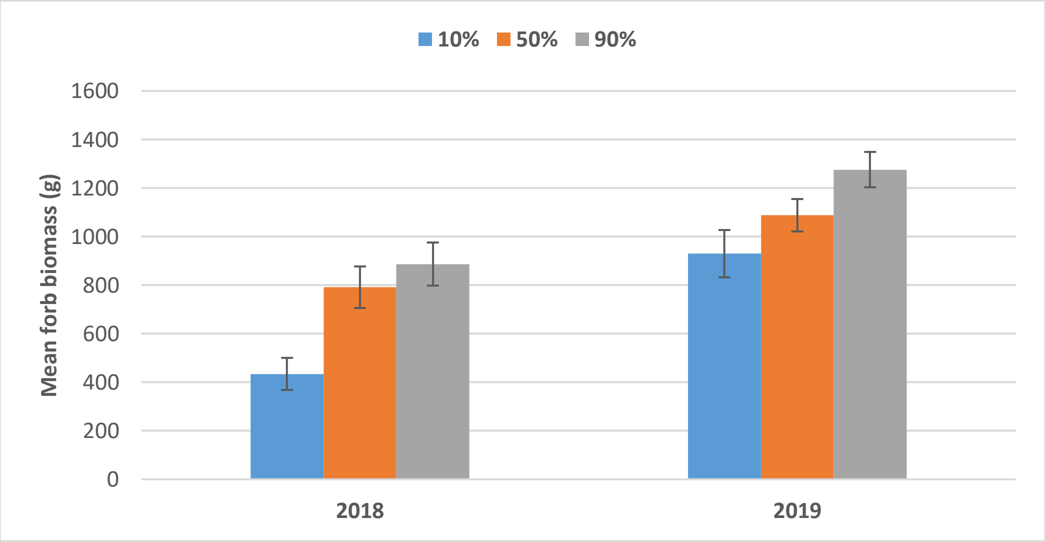


Figure 5.11 Effect of sowing ratio of forb: grass on forb biomass/ plot in August 2018 and August 2019 (Error bar = 2 Standard Errors)

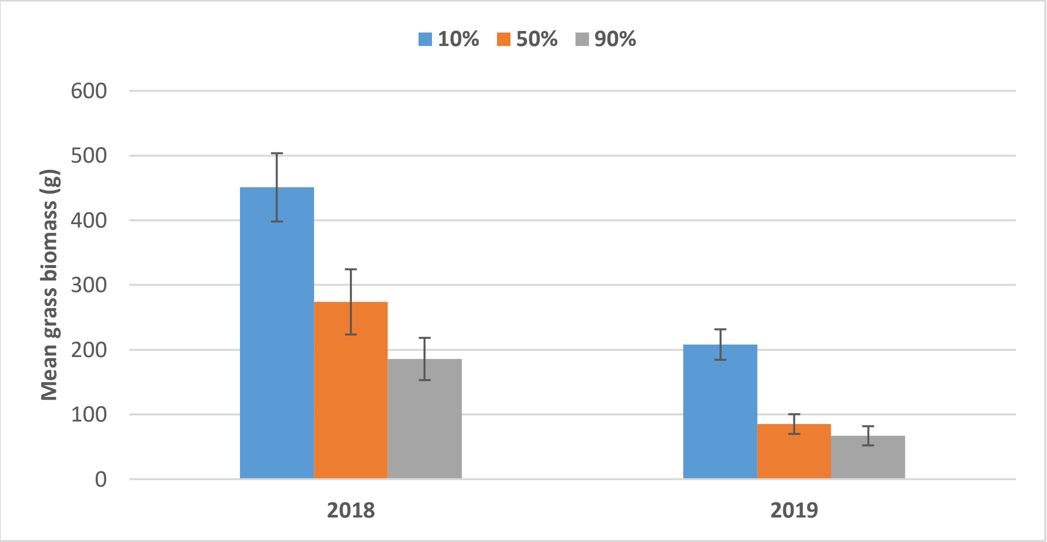


Figure 5.12 Effect of sowing ratio of forb: grass on grass biomass/ plot in August 2018 and August 2019 (Error bar = 2 Standard Errors)

iii) Forb, grass and bare ground cover values

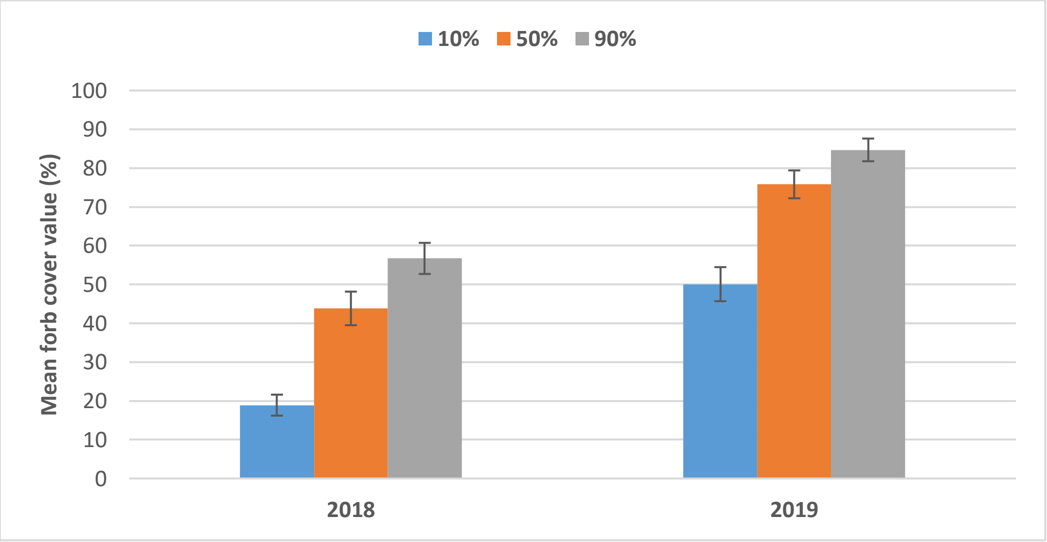


Figure 5.13 Effect of sowing ratio of forb: grass on forb cover values in May 2018 and May 2019 (Error bar = 2 Standard Errors)

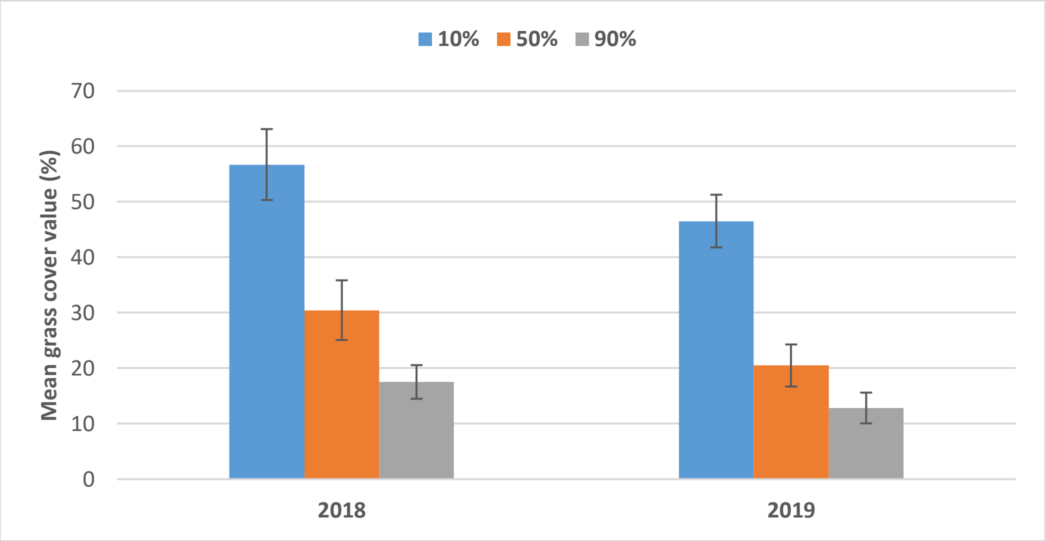


Figure 5.14 Effect of sowing ratio of forb: grass on grass cover values in May 2018 and May 2019 (Error bar = 2 Standard Errors)

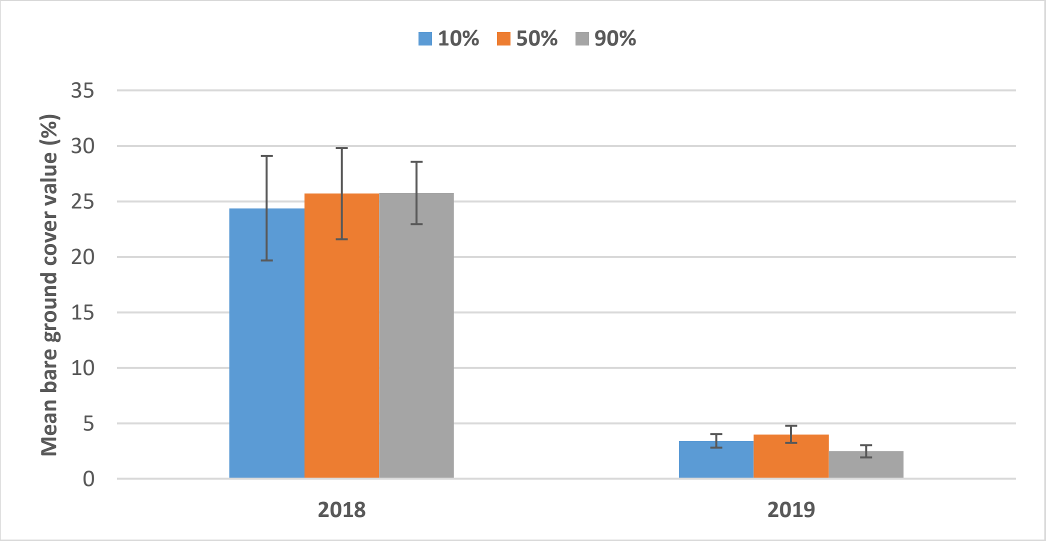


Figure 5.15 Effect of sowing ratio of forb: grass on bare ground cover values in May 2018 and May 2019 (Error bar = 2 Standard Errors)

5.3.1.2 Effect of sowing ratio of forb: grass on dominant forbs (*Achillea millefolium* and *Echinops sphaerocephalus*) and subordinate forb biomass in 2018 and 2019

Increasing the sowing ratio of forb: grass from 1: 9 to 1: 1 significantly increased biomass of *Achillea millefolium* in both years (p=0.000) (as shown in Figure 5.16). However, a further increase in the ratio from 1: 1 to 9:1 did not significantly increase biomass of *A. millefolium* from 2018 (p=0.292 in 2018 and p=0.512 in 2019). Interestingly, the 50% treatment had more *A. millefolium* biomass (726.47g) than the 90% (683.72g) in 2019 although the difference was small.

Sowing ratio of forb: grass did not significantly affect the biomass of *Echinops sphaerocephalus* in 2018 (p=0.705 between the 10% and 50% treatments and p=0.725 between the 50% and 90% treatment) (as shown in Figure 5.17). However, the pattern dramatically changed in 2019. The 10% and 90% treatment had similar amount of *E. sphaerocephalus* biomass (p=0.916) that were significantly more than the biomass in the 50% treatment (p=0.027 between the 10% and 50%, and p=0.009 between the 50% and 90% treatment). This pattern appeared irrespective to the original sowing density.

In 2018, subordinate forb biomass significantly increased from the ratio of 1: 9 to 1: 1 (p=0.015) but no significant difference between the 50% and 90% treatment (p=0.694) (as shown in Figure 5.18). Sowing more forb seed and reducing grass seeds did not increase subordinate forb biomass beyond that achieved in the forb to grass ratio of 1: 1. This was mostly because that the subordinate forb biomass was suppressed by both grass and dominant forb competition. In 2019, subordinate forb biomass had no significant difference between all treatments (p=0.112 between the 10% and 50%, p=0.553 between the 10% and 90%, and p=0.468 between the 50% and 90% treatment). However, the 50% treatment produced the highest amount of subordinate biomass (111.26g comparing with 81.66g in the 10% treatment and 91.92g in the 90% treatment).

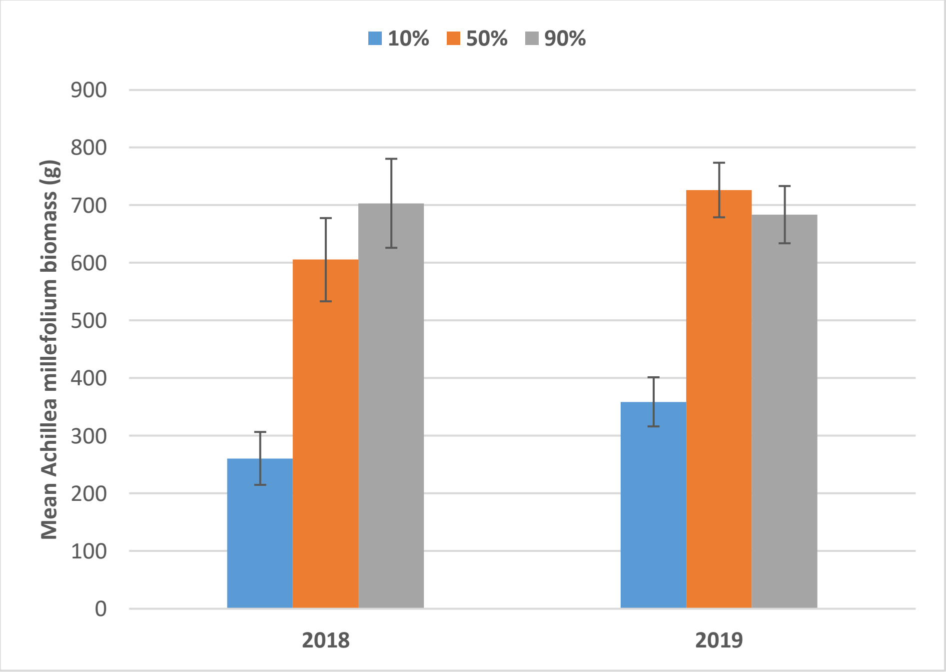


Figure 5.16 Effect of sowing ratio of forb: grass on *Achillea millefolium* biomass/ plot in August 2018 and August 2019 (Error bar = 2 Standard Errors)

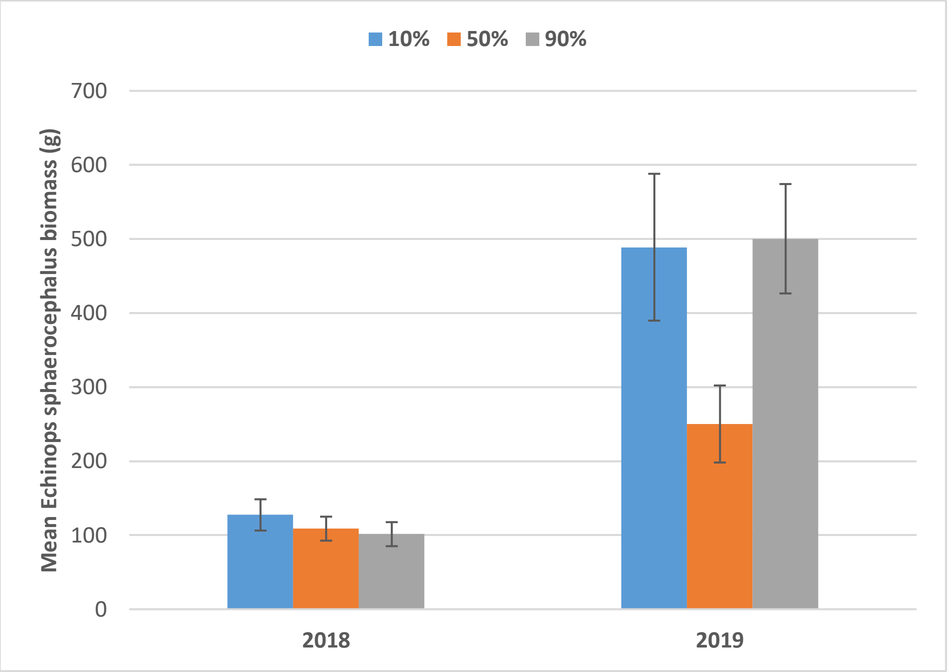


Figure 5.17 Effect of sowing ratio of forb: grass on *Echinops sphaerocephalus* biomass/ plot in August 2018 and August 2019 (Error bar = 2 Standard Errors)

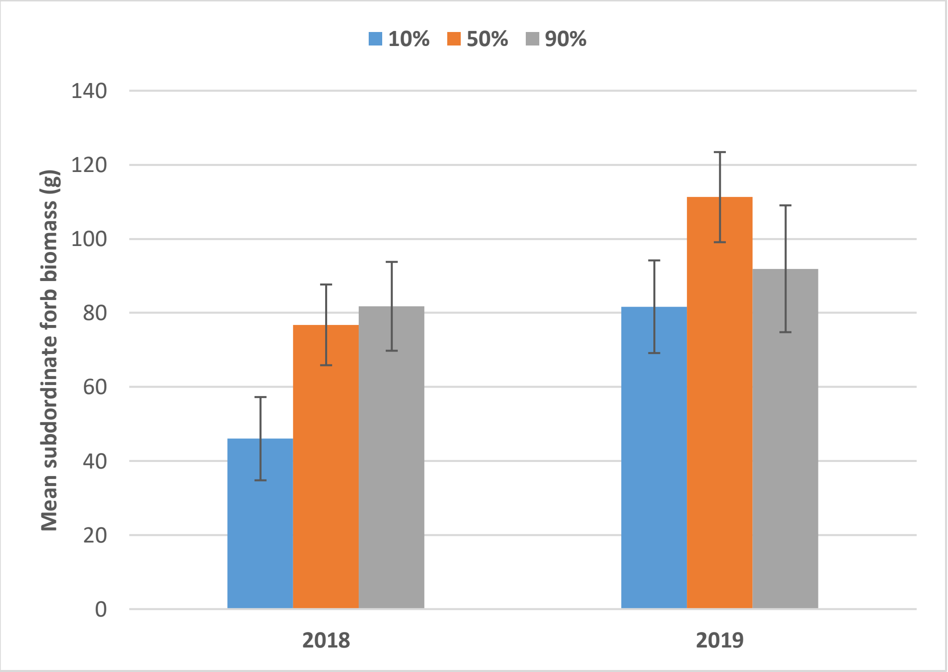


Figure 5.18 Effect of sowing ratio of forb: grass on subordinate forb biomass/ plot in August 2018 and August 2019 (Error bar = 2 Standard Errors)

Table 5.4 Effect of sowing ratio of forb: grass on overall plot results in 2018 and 2019 (SE = Standard Error of Mean)

						2018			
	10% forb		**50% forb**		**90% forb**		*P* value (between 10% and 50%)	*P* value (between 10% and 90%)	*P* value (between 50% and 90%)
	Mean	SE	Mean	SE	Mean	SE			
Forb seedling present number	71.59	7.04	242.47	19.32	360.56	26.52	0.000 ***	0.000 ***	0.000 ***
Forb biomass (g)	434.23	66.07	791.15	85.68	886.58	88.80	0.000 ***	0.000 ***	0.382 ns
Grass biomass (g)	450.93	52.74	273.97	50.38	185.79	32.64	0.002 **	0.000 ***	0.034 *
Forb cover value (%)	18.91	2.70	43.84	4.33	56.72	4.01	0.000 ***	0.000 ***	0.007 **
Grass cover value (%)	56.72	6.39	30.47	5.38	17.53	3.03	0.000 ***	0.000 ***	0.003 **
Bare ground cover value (%)	24.38	4.71	25.69	4.11	25.75	2.81	0.987 ns	0.987 ns	0.987 ns
***Achillea millefolium* biomass (g)**	260.84	45.90	605.56	72.21	703.41	77.21	0.000 ***	0.000 ***	0.292 ns
***Echinops sphaerocephalus* biomass (g)**	127.36	21.11	108.82	16.19	101.39	16.24	0.705 ns	0.680 ns	0.725 ns
Subordinate forb biomass (g)	46.03	11.24	76.77	10.90	81.78	12.00	0.015 *	0.015 *	0.694 ns

						2019			
	10% forb		**50% forb**		**90% forb**		*P* value (between 10% and 50%)	*P* value (between 10% and 90%)	*P* value (between 50% and 90%)
	Mean	SE	Mean	SE	Mean	SE			
Forb seedling present number	56.22	5.49	129.09	7.98	188.53	11.74	0.000 ***	0.000 ***	0.000 ***
Forb biomass (g)	929.51	97.29	1087.86	66.75	1275.93	72.77	0.138 ns	0.006**	0.083 ns
Grass biomass (g)	208.06	23.55	85.25	15.23	67.08	14.82	0.000 ***	0.000 ***	0.340 ns
Forb cover value (%)	50.09	4.39	75.81	3.59	84.69	2.92	0.000 ***	0.000 ***	0.030 *
Grass cover value (%)	46.53	4.75	20.50	3.79	12.84	2.77	0.000 ***	0.000 ***	0.063 ns
Bare ground cover value (%)	3.41	0.61	4.00	0.77	2.47	0.55	0.509 ns	0.389 ns	0.203 ns
***Achillea millefolium* biomass (g)**	358.99	42.63	726.47	47.36	683.72	49.66	0.000 ***	0.000 ***	0.512 ns
***Echinops sphaerocephalus* biomass (g)**	488.87	99.16	250.13	52.03	500.30	73.84	0.027 *	0.916 ns	0.009 **
Subordinate forb biomass (g)	81.66	12.51	111.26	12.17	91.92	17.13	0.112 ns	0.553 ns	0.468 ns

5.3.1.3 Effect of sowing ratio of forb: grass on forb seedling number per species in 2018 and 2019

Increasing sowing ratio of forb: grass significantly increased seedling numbers for most of the species in both 2018 and 2019 (refer to Table 5.5). Almost all forb species had significant increases in seedling number from the 10% to 50% and from the 10% to 90% forb treatment in 2018 with the exception of *Veronica incana* (p=0.488 between the 10% and 50% treatment). However, a few species had no significant increase in seedling numbers between the 50% and 90% treatment in both 2018 and 2019; *Anemone sylvestris* (p=0.555 in 2018, p=0.343 in 2019), *Pulsatilla vulgaris* (p=0.382 in 2018, p=0.827 in 2019), *Patrinia scabiosifolia* (p=0.378 in 2018, p=0.718 in 2019), *Echinops ritro* (p=0.351 in 2018, p=0.782 in 2019) and *Veronica longifolia* (p=0.152 in 2018, p=0.932 in 2019).

Increasing sowing ratio of forb: grass had no significant difference in all comparisons of the treatments on seedling numbers in 2019 for *Veronica incana*, *Platycodon grandiflorum, Scutellaria baicalensis, Patrinia scabiosifolia, Echinops ritro, Veronica longifolia* and *Angelica sylvestris*. *Thermopsis lanceolata* and *Dracocaphalum* ruychiana showed no significant differences between the 50% and 90% treatment and *Echinops sphaerocephalus* had no significant difference between the 10% and 50% treatment in 2019.

Table 5.5 Effect of sowing ratio of forb: grass on seedling number of each forb species in 2018 and 2019 (SE = Standard Error of Mean; na = not applicable)

	2018								
	10% forb		**50% forb**		**90% forb**		*P* value (between 10% and 50%)	*P* value (between 10% and 90%)	*P* value (between 50% and 90%)
	Mean	SE	Mean	SE	Mean	SE			
Low canopy									
Shared									
Anemone sylvestris	2.63	0.446	10.38	1.647	11.38	1.782	0.000 ***	0.000 ***	0.555 ns
Galium verum	2.38	0.372	9.34	1.389	16.19	2.082	0.000 ***	0.000 ***	0.001 **
Potentilla rupestris	3.72	0.613	9.84	0.982	15.50	1.544	0.000 ***	0.000 ***	0.001 **
Pulsatilla vulgaris	2.28	0.389	9.97	1.576	10.44	1.869	0.000 ***	0.000 ***	0.382 ns
Veronica teucrium	3.34	0.527	9.06	1.405	13.66	1.923	0.000 ***	0.000 ***	0.037 *
Mongolian									
Dracocephalum rupestre	3.91	0.528	17.75	1.861	27.66	2.267	0.000 ***	0.000 ***	0.000 ***
Dracocephalum ruychiana	2.22	0.386	13.47	2.128	19.22	2.642	0.000 ***	0.000 ***	0.002 **
Thalictrum petaloideum	3.88	0.470	13.00	1.284	17.31	1.661	0.000 ***	0.000 ***	0.008 **
Thermopsis lanceolata	1.72	0.351	9.50	1.367	13.94	1.939	0.000 ***	0.000 ***	0.003 **
Veronica incana	0.13	0.059	0.19	0.083	1.00	0.460	0.488 ns	0.018 *	0.034 *
Medium canopy									
Shared									
Achillea millefolium	1.66	0.132	6.47	0.460	11.72	0.737	0.000 ***	0.000 ***	0.000 ***
Origanum vulgare	11.25	1.337	28.41	2.834	41.66	3.799	0.000 ***	0.000 ***	0.001 **
Polemonium caeruleum	1.47	0.229	8.53	1.033	11.78	1.022	0.000 ***	0.000 ***	0.001 **
Stachys officinalis	1.50	0.090	6.50	0.449	12.00	0.718	0.000 ***	0.000 ***	na
Mongolian									
Campanula punctata	9.59	1.690	24.41	3.426	35.75	3.499	0.000 ***	0.000 ***	0.001 **
Delphinium grandiflorum	3.22	0.599	13.84	1.737	21.66	2.374	0.000 ***	0.000 ***	0.000 ***
Kalimeris incisa	2.41	0.373	9.97	0.984	18.69	1.790	0.000 ***	0.000 ***	0.000 ***
Platycodon grandiflorum	0.81	0.217	4.22	0.726	7.97	1.331	na	na	na
Scutellaria baicalensis	0.56	0.195	1.63	0.396	4.47	0.871	na	na	na
Tall canopy									
Shared									
Echinops ritro	0.63	0.147	1.56	0.294	2.25	0.563	0.007 **	0.005 **	0.351 ns
Geranium pratense	2.41	0.364	8.06	0.721	8.78	0.728	0.000 ***	0.000 ***	0.217 ns
Sanguisorba officinalis	0.47	0.135	2.38	0.294	4.16	0.638	0.000 ***	0.000 ***	0.015 *
Thalictrum aquilegifolium	2.00	0.280	8.59	0.783	12.09	1.333	0.000 ***	0.000 ***	0.008 **
Veronica longifolia	1.28	0.331	2.94	0.548	4.34	0.867	0.008 **	0.001 **	0.152 ns
Mongolian									

Aconitum carmichaelii	1.00	0.000	1.00	0.000	1.00	0.000	na	1.000 ns	1.000 ns
Angelica sylvestris	1.00	0.000	2.50	0.090	4.00	0.180	0.000 ***	na	na
Echinops sphaerocephalus	2.84	0.250	4.28	0.255	7.28	0.438	0.000 ***	0.000 ***	0.000 ***
Patrinia scabiosifolia	1.31	0.208	4.69	0.650	4.69	0.457	0.000 ***	0.000 ***	0.378 ns

2019

	10% forb		**50% forb**		**90% forb**		*P* value (between 10% and 50%)	*P* value (between 10% and 90%)	*P* value (between 50% and 90%)
	Mean	SE	Mean	SE	Mean	SE			
Low canopy									
Shared									
Anemone sylvestris	2.38	0.481	5.97	1.055	6.53	1.049	0.001 **	0.000 ***	0.343 ns
Galium verum	2.13	0.338	7.03	0.867	11.50	1.276	0.000 ***	0.000 ***	0.001 **
Potentilla rupestris	3.38	0.564	6.75	0.543	10.81	0.992	0.000 ***	0.000 ***	0.000 ***
Pulsatilla vulgaris	1.03	0.264	3.84	0.900	3.16	0.741	0.009 **	0.016 *	0.827 ns
Veronica teucrium	3.06	0.577	5.69	0.805	9.50	1.249	0.002 **	0.000 ***	0.002 **
Mongolian									
Dracocephalum rupestre	1.22	0.300	4.91	0.757	8.44	1.141	0.000 ***	0.000 ***	0.002 **
Dracocephalum ruychiana	0.69	0.235	3.16	0.594	3.41	0.766	0.000 ***	0.000 ***	0.375 ns
Thalictrum petaloideum	2.63	0.455	6.00	0.727	9.25	1.069	0.000 ***	0.000 ***	0.001 **
Thermopsis lanceolata	0.75	0.168	2.19	0.482	2.75	0.544	0.021 *	0.000 ***	0.607 ns
Veronica incana	0.00	0.000	0.00	0.000	0.00	0.000	0.779 ns	0.428 ns	0.779 ns
Medium canopy									
Shared									
Achillea millefolium	1.66	0.132	6.47	0.460	11.72	0.737	0.000 ***	0.000 ***	0.000 ***
Origanum vulgare	9.31	1.075	15.56	1.305	22.13	1.510	0.000 ***	0.000 ***	0.000 ***
Polemonium caeruleum	0.75	0.215	3.66	0.553	6.00	1.150	0.000 ***	0.000 ***	0.018 *
Stachys officinalis	1.09	0.122	3.41	0.342	7.91	0.563	0.000 ***	0.000 ***	0.000 ***
Mongolian									
Campanula punctata	9.09	2.083	15.03	2.275	19.19	2.195	0.007 **	0.000 ***	0.022 *
Delphinium grandiflorum	2.97	0.767	10.22	1.961	12.34	1.323	0.000 ***	0.000 ***	0.002 **
Kalimeris incisa	1.47	0.354	5.25	0.674	9.22	1.065	0.000 ***	0.000 ***	0.000 ***
Platycodon grandiflorum	0.00	0.000	0.00	0.000	0.09	0.094	1.000 ns	1.000 ns	1.000 ns
Scutellaria baicalensis	0.03	0.031	0.28	0.121	0.25	0.090	1.000 ns	1.000 ns	1.000 ns
Tall canopy									
Shared									
Echinops ritro	0.34	0.096	0.84	0.211	1.16	0.404	0.113 ns	0.158 ns	0.782 ns
Geranium pratense	3.88	0.685	6.75	0.808	9.47	0.932	0.001 **	0.000 ***	0.003 **
Sanguisorba officinalis	0.47	0.135	2.03	0.260	3.50	0.389	0.000 ***	0.000 ***	0.002 **
Thalictrum aquilegifolium	2.28	0.255	6.47	0.589	9.31	0.858	0.000 ***	0.000 ***	0.002 **
Veronica longifolia	1.69	0.452	2.72	0.601	2.63	0.628	0.493 ns	0.493 ns	0.932 ns
Mongolian									
Aconitum carmichaelii	0.81	0.070	0.81	0.070	0.88	0.059	0.873 ns	0.740 ns	0.740 ns
Angelica sylvestris	0.59	0.099	0.56	0.142	0.72	0.150	0.933 ns	0.933 ns	0.933 ns
Echinops sphaerocephalus	2.22	0.335	3.16	0.484	6.06	0.502	0.174 ns	0.000 ***	0.000 ***
Patrinia scabiosifolia	0.31	0.138	0.34	0.124	0.63	0.470	0.542 ns	0.718 ns	0.718 ns

5.3.1.4 Effect of sowing ratio of forb: grass on subordinate forb biomass per species in 2018 and 2019

Increasing sowing ratio of forb: grass did not increase the biomass for most of the subordinate forb species in both 2018 and 2019 especially for the low canopy forbs (refer to Table 5.6). In 2018, *Stachys officinalis* was the only species that increased biomass with the higher sowing ratio of forb: grass (p=0.000). *Dracocephalum rupestre* (p=0.006 and p=0.000), *Kalimeris incisa* (p=0.001 and p=0.000) and *Thalictrum aquilegifolium* (p=0.004 and p=0.000) significantly increased in biomass from the 10% to 50% and the 10% to 90% forb treatment. *Sanguisorba officinalis* had significantly greater biomass increase from the 10% to the 90% forb treatment (p=0.008). Interestingly, *Geranium pratense* had significantly more biomass in the 50% treatment than the 10% (p=0.002 in 2018 and p=0.037 in 2019) and 90% treatment (p=0.022 in 2018 and p=0.043 in 2019) in both 2018 and 2019.

In 2019, *Stachys officinalis* (p=0.002), *Sanguisorba officinalis* (p=0.002) and *Thalictrum aquilegifolium* (p=0.003) still had more biomass in the 90% treatment than the 10% treatment. *Scutellaria baicalensis* (p=0.038) and *Echinops ritro* (p=0.030) had significantly more biomass in the 50% treatment than the 90% treatment.

Table 5.6 Effect of sowing ratio of forb: grass on biomass of each subordinate forb species in 2018 and 2019 (SE = Standard Error of Mean; na = not applicable)

	2018								
	10% forb		**50% forb**		**90% forb**		*P* value (between 10% and 50%)	*P* value (between 10% and 90%)	*P* value (between 50% and 90%)
	Mean	SE	Mean	SE	Mean	SE			
Low canopy									
Shared									
Anemone sylvestre	0.08	0.04	0.09	0.03	0.06	0.02	0.815 ns	0.815 ns	0.617 ns
Galium verum	1.24	0.59	4.48	1.72	3.32	1.03	0.062 ns	0.132 ns	0.480 ns
Potentilla rupestris	3.14	1.51	2.60	0.52	3.33	0.81	0.908 ns	0.908 ns	0.772 ns
Pulsatilla vulgaris	0.09	0.06	0.21	0.06	0.17	0.06	0.374 ns	0.568 ns	0.584 ns
Veronica teucrium	1.35	0.66	1.07	0.38	1.52	0.35	0.884 ns	0.884 ns	0.701 ns
Mongolian									
Dracocephalum rupestre	0.99	0.31	2.54	0.52	3.63	0.56	0.006 **	0.000 ***	0.084 ns
Dracocephalum ruychiana	0.67	0.23	0.78	0.20	1.15	0.25	0.668 ns	0.267 ns	0.321 ns
Thalictrum petaloideum	0.00	0.00	0.00	0.00	0.00	0.00	1.000 ns	1.000 ns	na
Thermopsis lanceolata	0.79	0.30	1.54	0.59	1.20	0.29	0.529 ns	0.529 ns	0.581 ns
Veronica incana	0.00	0.00	0.00	0.00	0.00	0.00	0.854 ns	0.854 ns	0.883 ns
Medium canopy									
Shared									
Origanum vulgare	14.19	4.24	16.54	2.85	23.25	4.91	0.585 ns	0.215 ns	0.257 ns
Polemonium caeruleum	1.78	1.25	1.29	0.52	0.78	0.25	0.710 ns	0.710 ns	0.710 ns
Stachys officinalis	0.08	0.02	0.30	0.06	0.82	0.13	0.000 ***	0.000 ***	0.000 ***
Mongolian									

Campanula punctata	2.09	0.79	2.30	0.84	1.38	0.38	0.835 ns	0.588 ns	0.586 ns
Delphinium grandiflorum	2.46	0.92	3.52	1.21	1.30	0.37	0.409 ns	0.381 ns	0.105 ns
Kalimeris incisa	7.70	2.56	21.42	4.65	27.86	5.75	0.001 **	0.000 ***	0.274 ns
Platycodon grandiflorum	0.08	0.03	0.07	0.02	0.11	0.02	0.762 ns	0.632 ns	0.382 ns
Scutellaria baicalensis	0.38	0.19	0.95	0.39	0.51	0.16	0.342 ns	0.559 ns	0.430 ns
Tall canopy									
Shared									
Echinops ritro	3.28	1.71	5.76	1.76	2.29	0.95	0.481 ns	0.594 ns	0.193 ns
Geranium pratense	1.90	0.45	6.02	1.25	2.78	0.62	0.002 **	0.227 ns	0.022 *
Sanguisorba officinalis	1.08	0.52	3.16	1.30	3.82	0.84	0.215 ns	0.008 **	0.652 ns
Thalictrum aquilegifolium	0.02	0.01	0.08	0.02	0.12	0.02	0.004 **	0.000 ***	0.089 ns
Veronica longifolia	1.97	1.03	1.10	0.39	1.44	0.78	0.737 ns	0.875 ns	0.875 ns
Mongolian									
Aconitum carmichaelii	0.20	0.05	0.23	0.05	0.34	0.06	0.699 ns	0.162 ns	0.203 ns
Angelica sylvestris	0.04	0.03	0.01	0.00	0.04	0.01	0.340 ns	0.978 ns	0.011 *
Patrinia scabiosifolia	0.42	0.15	0.72	0.22	0.57	0.21	0.456 ns	0.717 ns	0.717 ns

	2019								
	10% forb		**50% forb**		**90% forb**		*P* value (between 10% and 50%)	*P* value (between 10% and 90%)	*P* value (between 50% and 90%)
	Mean	SE	Mean	SE	Mean	SE			
Low canopy									
Shared									
Anemone sylvestre	0.11	0.06	0.03	0.01	0.01	0.00	0.241 ns	0.167 ns	0.241 ns
Galium verum	4.35	1.38	10.38	2.99	8.19	1.63	0.064 ns	0.109 ns	0.441 ns
Potentilla rupestris	5.90	3.79	3.56	0.78	3.13	1.25	0.820 ns	0.820 ns	0.820 ns
Pulsatilla vulgaris	0.03	0.02	0.08	0.04	0.02	0.01	0.461 ns	0.643 ns	0.411 ns
Veronica teucrium	2.97	1.07	1.22	0.33	1.74	0.48	0.147 ns	0.345 ns	0.345 ns
Mongolian									
Dracocephalum rupestre	0.08	0.04	0.12	0.05	0.09	0.03	0.855 ns	0.855 ns	0.855 ns
Dracocephalum ruychiana	0.34	0.28	0.18	0.09	0.26	0.17	0.912 ns	0.912 ns	0.912 ns
Thalictrum petaloideum	0.00	0.00	0.00	0.00	0.00	0.00	0.562 ns	0.562 ns	0.709 ns
Thermopsis lanceolata	0.58	0.28	0.48	0.26	0.30	0.10	0.765 ns	0.663 ns	0.753 ns
Veronica incana	0.00	0.00	0.00	0.00	0.00	0.00	na	na	1.000 ns
Medium canopy									
Shared									
Origanum vulgare	40.96	7.29	43.38	5.98	41.91	9.67	0.988 ns	0.988 ns	0.988 ns
Polemonium caeruleum	1.23	0.86	1.10	0.39	1.32	0.45	0.984 ns	0.984 ns	0.964 ns
Stachys officinalis	0.45	0.18	1.37	0.70	1.67	0.32	0.309 ns	0.002 **	0.676 ns
Mongolian									
Campanula punctata	1.73	0.95	1.00	0.48	0.39	0.17	0.446 ns	0.337 ns	0.337 ns
Delphinium grandiflorum	2.98	1.32	2.50	1.14	0.81	0.46	0.762 ns	0.278 ns	0.278 ns
Kalimeris incisa	4.95	1.87	12.26	4.27	14.39	4.72	0.129 ns	0.102 ns	0.701 ns
Platycodon grandiflorum	0.01	0.01	0.01	0.00	0.01	0.00	0.780 ns	0.780 ns	0.780 ns
Scutellaria baicalensis	0.11	0.07	0.14	0.06	0.01	0.01	0.735 ns	0.213 ns	0.038 *
Tall canopy									
Shared									
Echinops ritro	2.52	1.44	6.30	2.18	0.85	0.30	0.245 ns	0.245 ns	0.030 *

Geranium pratense	8.09	1.72	21.33	5.67	8.86	2.01	0.037 *	0.756 ns	0.043 *
Sanguisorba officinalis	1.41	0.67	4.04	1.72	6.41	1.36	0.238 ns	0.002 **	0.251 ns
Thalictrum aquilegifolium	0.01	0.00	0.07	0.04	0.06	0.01	0.138 ns	0.003 **	0.661 ns
Veronica longifolia	1.55	0.67	1.35	0.54	0.95	0.39	0.805 ns	0.802 ns	0.802 ns
Mongolian									
Aconitum carmichaelii	0.34	0.11	0.26	0.07	0.38	0.11	0.784 ns	0.784 ns	0.635 ns
Angelica sylvestris	0.86	0.78	0.02	0.02	0.07	0.03	0.564 ns	0.564 ns	0.677 ns
Patrinia scabiosifolia	0.10	0.04	0.07	0.04	0.08	0.03	0.935 ns	0.935 ns	0.935 ns

5.3.2 Discussion on effects of sowing ratio of forb: grass

5.3.2.1 The effect of ratios of forb: grass on overall number of forb seedlings, biomass of forb and grass and cover values

Increasing the sowing ratio of forb: grass gets more forb seedlings in both 2018 and 2019 (p=0.000 for all) but the increase was not linear. There was no significant difference in abundance of forb biomass unless a distinctive gap had been made on the ratios especially in the third year (2019) (no significant difference between the 50% to the 90% treatment in 2018, p=0.382; no significant difference between both 10% to 50%, p=0.138 and 50% to 90% treatment, p=0.083). This suggests that the higher seed ratio of forb: grass in more forb seedlings for at least the first three years. This agrees with Hitchmough et al., (2017) that altering the seed ratio within a mix affects the numbers of seedling but not the biomass. This effect gradually diminished in the longer term. In addition, since the forb seeds are usually more expensive than the grass seeds, increasing the seed ratio of forb: grass may not be cost-effective beyond a certain threshold on the ratio in the longer term (Meissen et al, 2017). It may be a significant time for all forb performances to become similar in terms of seedling numbers and biomass production, across all the treatments given the hugely different starting point, however the trend of diminishment in forb numbers seems unavoidable. However, the value of having a high ratio of forb: grass can delay the loss of species richness and provide opportunities through further management approaches to restrict dominance.

The higher ratio of forb: grass led to a reduced increase in both forb seedling numbers and production of biomass. In 2018, the 10% treatment had 72 forb seedlings per plot whereas only 49 more (per 10% forb sowing ratio increase) forb seedlings were added in the 50% treatment and a further 41 more forb seedlings in the 90% treatment. These constraints were greater for the forb biomass; the 10% treatment had 434.23g per plot which went down to 158.23g and 98.51g more (per 10% forb sowing ratio increase) in the 50% and 90% treatments. In 2019, the effect of the ratios was further diminished that while there were 56 forb seedlings in the 10% treatment, only 26 and 21 more (per 10% forb sowing ratio increase) forb individuals were added in the 50% and 90% treatment. For the forb biomass, the 10% treatment produced 929.51g but only 217.57g and 141.77g more (per 10% forb sowing ratio increase) in the 50% and 90% treatments. In terms of the seedling number, the 90% treatment led to the highest forb mortality. This pattern can be explained by the self-thinning rule (Yoda *et al.*, 1963). The higher ratio of forb: grass initially led to a higher presence of forb seedlings with a greater amount of biomass in the early stage. Also, the top layer, dominant by the vigorous species such as *Achillea millefolium* and *Echinops sphaerocephalus*, or other species which developed tall leaves in the early stage, were more closed earlier due to a greater seedling number. This caused greater light extinction to the rest of forbs and likely eliminated them.

The grass competition might also lead to a substantial amount of the forb elimination in the 90% treatment. The grasses followed a similar pattern to the forb biomass. 185.79g of grass biomass was produced in the 90% (forb) treatment whereas 54.79g and 50.10g were produced in the 50% (forb) and 10% (forb) treatment in 2018. The figures decreased to 67.08g, 17.05g and 23.12g respectively (per 10% grass sowing ratio increase), but still showed similar trends. It suggests that the growth of grass was not linearly decreased with the designed ratio of forb: grass and this mostly due to the intra species competition between grasses. Despite that the competitive forbs could limit the growth of grasses especially in 2019, the grasses were still more competitive than most of the forb species. After all, the largest loss in the number of forbs was likely because of both vigorous forb competition and grass competition in the 90% (forb) treatment.

In terms of forb biomass, the lowest ratio of forb: grass (the 10% treatment) led to the highest value of forb biomass: sowing rate. This maximised the benefit: cost ratio, and can be explained by the facts that the vigorous forbs (i.e. *Achillea millefolium* and *Echinops sphaerocephalus*) within the mix, which inevitably contribute the most of forb biomass, can hence can overcome grass competition, despite the seedling density was low.

In 2018, there was no significant difference on the forb biomass between the 50% and 90% treatments. This is probably because the forb biomass pattern in summer was mainly driven by the dominant species and those species that appeared less affected by the grass competition in those treatments. Despite the 90% treatment allowed forbs to reach a higher coverage in spring (44% in the 50% treatment and 57% in the 90% treatment, p=0.007), it potentially just ensured that the biomass ceiling was reached sooner but this led to a greater intra and inter species competition mainly between the dominant forbs. Forbs reached a similar level of biomass in the 50% treatment but just took a longer time. This applies to precedent studies that indicate the structure of biomass production are irrespective to the original sowing ratios between the species, but dominant species are inevitably advantageous. A lower seed density would be sufficient to create a similar level of establishment in the longer term (Stevenson, Bullock and Ward, 1995; Hitchmough, Wagner and Ahmad, 2017). However, the 10% treatment still led to a semi-grass dominated community from spring (as 57% of grass coverage) to the summer (as the ratio of forb: grass biomass = 1: 1.04). This was ascribed to both extremely high grass seeds and low forb seeds input.

In 2019, the overall reduction in the grass biomass led to a significant increase on the forb biomass in all treatments (p=0.000). The significant difference of forb biomass only occurred between the two extreme conditions (10% and 90% treatment) (p=0.006). Despite that the grass biomass in the 10% treatment was still significantly higher than the 50% and 90% treatment (p=0.000), the community appeared as forb dominant in summer in 2019.

5.3.2.2 The effect of ratio of forb: grass on biomass of dominants (*Achillea millefolium* and *Echinops sphaerocephalus*) and subordinates

The biomass production of *Achillea millefolium* appeared similar in the 50% and 90% forb treatment in 2018 and 2019 (p=0.292 and p=0.512) but was significantly more than that in the 10% treatment in both years (p=0.000). The greater the number of individuals (p=0.000 in both years) should have produced more biomass but the increase was checked after reaching a certain threshold. The similar amount of biomass production in the 50% and the 90% treatment primarily suggests that the grass competition at this level seemed less effective than the growth *A. millefolium.* Intraspecies competition played a major role as the rhizomatous growth of *A. millefolium* led to a ceiling of biomass accumulation from 2018. This happened sooner in the 90% treatment due to significantly higher seedling numbers of *A. millefolium* than both the 10% and 50% treatments (p=0.000, and eventually made no significant difference in biomass to the 50% forb treatment). The accumulation of biomass can also lead to a self-thinning in the seedling number (Yoda *et al.*, 1963). The seedling numbers were presumably similar in both the 50% and the 90% treatment. However, there was no reliable seedling counting data in spring 2019 (as explained in Chapter 4) to support this assumption.

There was no biomass difference in the 90% forb treatment for *Achillea millefolium* between 2018 and 2019 (p=0.623). This suggests that *A. millefolium* had fully occupied the niches and reached the biomass ceiling in this treatment in 2018. *A. millefolium* outcompeted the grasses and had a significantly biomass increase in the 50% treatment between 2018 and 2019 (p=0.015). This suggests that a higher density of grass seeds in the mix could only delay the establishment of the *A. millefolium* community in the short term. It can be predicted that there will not be any big increase in the biomass of *A. millefolium* from 2019 due to the intra species competition. Only an extremely high grass presence with an extremely low number *A. millefolium* individuals (i.e. the 10% treatment), at about 1.7 *A. millefolium* seedlings per plot, at the initial stage could restrict the dominance for a longer period.

The biomass pattern of *Echinops sphaerocephalus* showed that a significantly different number of seedlings in spring (p=0.000) lead to a similar level of biomass production between each treatment in summer 2018 (p=0.705 between the 10% and 50%, p=0.680 between the 10% and 90% and p=0.725 between the 50% and the 90% treatment). This suggests that the grass competition was also less effective in restricting the growth the *E. sphaerocephalus.* On the other hand, *E. sphaerocephalus* could be suppressed by *A. millefolium* and potentially other competitive forbs especially at the seedling stage in spring 2018. Inter species competition was the main factor in reducing the seedling numbers. The intra species competition within *E. sphaerocephalus* could have taken place in summer when some individuals had developed large leaves but was believed as a less important factor since it did not play a significant role in dominance. This result indicates a trade-off between the effective growth of fewer individuals and the growth of more seedlings with a higher competitive stress. Besides, although there was no formal measurement, the observation indicates that the major amount of *E. sphaerocephalus* biomass was contributed by one or a few individuals which took the physical gap from the absence of vigorous forbs and gained the maximum growth while the rest of them remained small and could only add a relatively marginal amount of biomass in 2018. This again suggests a weak relation between the seedling number and biomass production of *E. sphaerocephalus* in 2018.

The biomass pattern of *Echinops sphaerocephalus* appeared different in 2019 that the 50% treatment had significantly lower biomass than the 10% and 90% treatments (p=0.027 between the 10% and 50%, p=0.916 between the 10% and 90% and p=0.009 between the 50% and 90% treatment). *A. millefolium* competition seemed to still play the major role to affect the biomass production of *E. sphaerocephalus.* Comparing the 10% with the 50% treatment, the former treatment had significantly low biomass of *A. millefolium* (p=0.000) which supported to a greater biomass of *E. sphaerocephalus.* Comparing the biomass of *E. sphaerocephalus* between the 50% and the 90% treatment, the most obvious causation to the greater biomass presence in the later treatment was the numerical advantage in the seedling (p=0.000) which could potentially find more chances to occupy the gap where had more resource availability under the similar level of *A. millefolium* competition (refer to the biomass production, p=0.512). Finally, as *E. sphaerocephalus* became dominant, the intra species competition is supposed as a more important factor to limit the biomass accumulation of *E. sphaerocephalus* in the 90% treatment, that had similar amount of biomass to the 10% treatment in 2019.

Having more forb seeds in the mix is unlikely to make the subordinate forbs more abundant in biomass in the longer term. The only significant difference occurred between the 10% (46.03g) treatment and the 50% (76.77g) treatment (p=0.015), as well as the 10% to 90% (81.78g) treatment (p=0.015) in 2018. This indicated that five times more forb seeds could only lead the subordinate biomass to increase by about 67%. In addition, the biomass volume appeared similar between the 50% treatment and the 90% treatment (p=0.694). This suggests that the threshold effectively inhibiting the increase of subordinate biomass might appear around the ratio of 1: 1 (forb: grass seeds). This can be explained by that the dominant forbs could break through the grass competition and accumulate similar amount of biomass beyond this point. Both grass and dominant forb competition inhibited the increase of subordinate biomass and kept them at a similar level between the 50% and 90% treatment. There was no significant difference on subordinate biomass between the treatments in 2019 (p=0.112 between the 10% and 50%, p=0.553 between the 10% and 90% and p=0.468 between the 50% and 90% treatment). Adding more forb seeds in the ratio could only increase the inter species competition from the dominants to the subordinates.

From 2018 to 2019, the 10% and 50% treatments had significant increases in subordinate forb biomass (p=0.000) with no significant increase in the 90% treatment (p=0.330). This suggested that the community where initially had more presence of grass could potentially encourage more growth of subordinate forbs when the grass biomass was reduced. The consistent presence of dominant forbs, especially *Achillea millefolium,* could inhibit the biomass accumulation of subordinate forbs.

Reducing seedling density of dominant species was a possible way to enhance subordinate biomass (Hitchmough and Fleur, 2006; Hitchmough and Wagner, 2013). Interactive results of ‘sowing density x sowing ratio of forb: grass’ (Figure 5.19 and Figure 5.20) showed increases of subordinate biomass from the forb to grass ratio of 1: 9 to 9: 1 in both 2018 and 2019 in the low density treatment (despite no significant difference except p=0.002 between the 10% and 90% treatment in 2018) but not the high density treatment. Also, low sowing density led to a significantly more subordinate biomass in the 90% treatment in 2019 (p=0.012 as shown in Figure 5.44). However, the author would argue that the most powerful design tool for a forb rich and diverse meadow community is not sowing density or ratio across all species but changing the ratio of seed of species for example A to B to C in particular reducing the dominant seeds to very low levels.

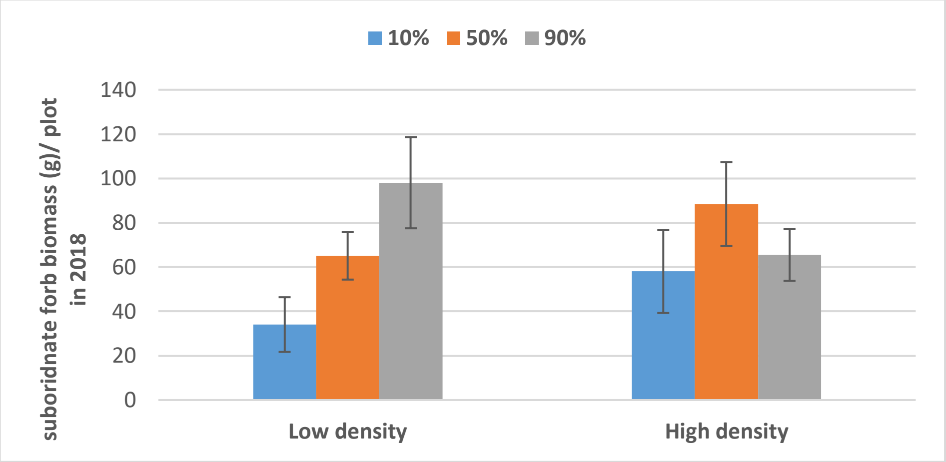


Figure 5.19 Interactive effect of 'sowing density x sowing ratio of forb: grass' on subordinate forb biomass in 2018 (Error bar = 2 Standard Errors)

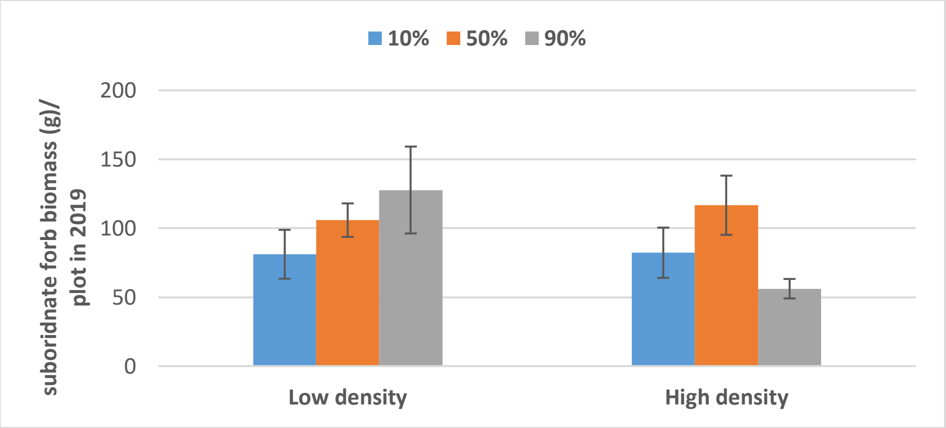


Figure 5.20 Interactive effect of 'sowing density x sowing ratio of forb: grass' on subordinate forb biomass in 2019 (Error bar = 2 Standard Errors)

5.3.2.3 The effect of ratios of forb: grass on the number of seedlings and biomass of subordinate species

Increasing the sowing ratio of forb: grass from the 1: 9 to 1: 1 was effective to increase the number of subordinate forbs for least three growing years. However, many forbs ceased to increase their number of seedlings beyond this point (i.e. from the ratio of 1: 1 to the ratio of 9: 1) from 2018. The general pattern suggests that increasing the sowing ratio of forb: grass more effectively increased the seedling numbers of the short and medium canopy forbs but less effective to the tall canopy forbs.

Within the low canopy forbs, seedling numbers of those with the characteristics of low rosette form appeared less affected by increasing sowing ratio of forb to grass from the 1: 1 to 9: 1 (i.e. *Anemone sylvestris*, p=0.555 between the 50% and 90% treatments; and *Pulsatilla vulgaris*, p=0.382 between the 50% and 90% treatments). However, the tall forbs (*Echinops* *ritro*, p=0.378 and *Veronica longifolia*, p=0.152) within the subordinate group with no statistical differences in terms of seedling number between the 50% and 90% treatment (from 2018) might be because of their upright growth form or tall leafy stems. Enhancing light competition might make them able to survive from grass competition but were less persistent to *Achillea millefolium* competition. *A. millefolium* competition involved rapid lateral expansion by the rhizomatous growth in the 90% forb treatment. Those species were probably less tolerant to the spatial competition.

Thermopsis lanceolata and *Dracocephalum ruychiana* had significantly more seedlings in the 90% than the 50% treatment in 2018 (p=0.003 and p=0.002) but there was no significant difference between the two treatment in 2019. This indicates that the competitive advantage to survive for the low canopy short leafy stem species may be transient. Moreover, *Echinops ritro* (p=0.113 between the 10% and 50% treatment and p=0.158 between the 10% and 90% treatment) and *Veronica longifolia* (p=0.493 between both the 10% and 50% treatment and p=0.493 the 10% and 90% treatment) showed no statistical difference between all the ratio treatments in 2019. The possible explanation for this may be that those species showed strong ability of shoot thrust or project tall leafy stems above the grass canopy to be survived from the 'grassy dominance' plots (the 10% treatment). However, when the dominance role was shifted to *A. millefolium* and *E. sphaeroephalus* dominance, their number of seedlings declined.

Sowing ratio can however enhance the survival of species which have characteristics of clambering stems (i.e. *Galium verum*), early emergence and growth (i.e. *Potentilla rupestris*) can enhance the possibility to survive with increased sowing ratio. For the tall forbs, elongating petioles and ability to quick recover (i.e. *Geranium pratense* and *Sanguisorba officinalis*) appeared to be essential to survive with the forb dominants of *A. millefolium* and *E. sphaeroephalus* in the long term. Species with shade tolerance can survive under dominants competition (i.e. *Thalictrum spp.*) (Hitchmough, Paraskevopoulou and Dunnett, 2008; Hitchmough, Wagner and Ahmad, 2017). It would be reasonable to predict that more species will have a similar level of seedling richness and biomass between the 50% and 90% treatment in the next few years and be significantly higher than the numbers in the 10% treatment.

Maintaining biomass production from increased sowing ratio of forb: grass requires access to light. The competitive species in forb: grass ratio treatments were typically the same species as in the 'sowing density' treatment. Tall leafy stem species of *Sanguisorba officinalis* (p=0.008 in 2018 and p=0.002 between the 10% and 90% treatment) and elongating stem species of *Geranium pratense* (p=0.002 and p=0.037 between the 10% and 50% treatment) tend to produce more biomass with increased sowing rates. Shade tolerant seedlings can also retain more biomass with increased number of seedlings (*Thalictrum aquilegifolium*, p=0.004 between the 10% and 50% treatment and p=0.000 between the 10% and 90% treatment). Moreover, it is worth noting that *Geranium pratense* produced significantly more biomass in the 50% treatment than the 90% treatment (p=0.022 in 2018 and p=0.043 in 2019). *Geranium pratense* biomass was likely to be more sensitive to *Achillea millefolium* competition than grass competition. *Echinops ritro* displayed the similar pattern despite the difference was not statistically different. Besides, despite the 90% accommodated a significantly more seedling numbers, many of them remained tiny and were below the size of biomass data capture (biomass harvested at the plant height above 20mm). For this reason, a large number of seedlings did not contribute to the biomass data collection.

To conclude, the overall pattern agrees with the findings in Hitchmough et al., (2017) that from the second year, the dominant species suppress the rest of species in the community irrespective to the original sowing. The start points of species ratio affect numerical abundance in a longer term but not their biomass. In this research architectural advantages for light competition seemed to be more helpful to coexist with grasses. However, to compete with dominant forbs of *Achillea millefolium*, both architectural advantage and early establishment are required but are unlikely to turn around the dominant pattern. Again, lowering density of dominant species may help to increase the richness of subordinates but increasing subordinate density merely leads to more elimination.

Chapter 6 Effects of grass height

6.1 Introduction

One of critical mechanisms by which grasses eliminate forb diversity and richness is through light competition by producing denser and taller tillers at different stages. Both grasses incorporated intentionally into meadows and weedy grass invasion has high likelihood to be present from an early stage in every meadow sowing project. However, including grass biomass in a designed mix is important for both functional and ecological values. The issue is therefore the sensitivity of forb richness and diversity to different height grass canopies within this process. Changes in grass height may dramatically affect the forb community and the overall appearance of the design, as asymmetric competition between species may only require a minor difference in the key traits, such as plant height.

In this chapter, the impact of two selections of *Deschampsia cespitosa* in the sowing mix and sowing density was assessed. The tall grass was represented 'Barcampsia' and the short grass by 'Pixie Fountain'. By having two selections of the same species, which were similar in most aspects other than height, and vigour it was hoped that more clarity would be possible in deciphering the results. The core question was does having different heights of grass species at the beginning affect forb performance in the longer term? The aim was to test the hypothesis that the taller grass is more likely to lead to a greater reduction in forb richness and diversity than the shorter grass. Also, by using the two height of grasses and having the gradient of grass competition, the result may suggest a design model of value in different aesthetic and ecological situations in future landscapes. However, due to the poor emergence and survival of *Deschampsia cespitosa* 'Barcampisa', the results in 2018 mostly reflected the effects of two different grass seedling densities and biomass. As a result of over sowing and planting 'Barcampsia' in 2018, a relatively uniform grass coverage between the two treatments was achieved in 2019. As a result, the tall grass half of the experiment is more akin to the effects of heights of grass on forb performance in the 'mid-term' of community establishment.

6.2 Results

6.2.1 Effect of grass height on forb seedling numbers, forb and grass biomass and cover values in 2018 and 2019

The tall grass treatment significantly increased the number of forb present seedlings in 2018 and 2019 (p=0.000) (as shown in Figure 6.1). This was mainly because tall grasses had lower emergence than short grasses in spring 2017 and 2018 (as discussed in 6.3 Discussion). Variations in grass competition to low seedling density greatly affected this result rather than the height of grass. The tall grass treatment also significantly increased forb biomass in 2018 (p=0.000). However, the pattern reversed in 2019 in that the short grass treatment had more forb biomass (1130.96g) than the tall grass treatment (1064.57g) despite the difference not being significant (p=0.435) (Figure 6.2).

The short grass biomass was approximately 2.3 times more than the tall grass biomass in 2018 (p=0.000). However, there was no significant difference in 2019 (p=0.438) but the tall grass had slightly more biomass (127.48g) than the short grass (112.77g) (as shown in Figure 6.3).

In terms of cover values (refer to Figure 6.4 – 6.6), similar patterns to the result of forb and grass biomass were shown, the short grass treatment led to significantly less forb coverage (32% in the short grass treatment comparing with 47% in the tall grass treatment, p=0.000) and significantly more grass coverage (48% in the short grass treatment comparing with 22% in the tall grass treatment, p=0.000) in 2018. However, there was no significant difference on cover values in 2019.

The short grass treatment significantly decreased bare ground coverage in 2018 (p=0.002) with no significant difference between the two treatment in 2019 (p=0.331). This indicates the effectiveness of a higher number of grass seedlings to close bare ground in the short term.

i) Forb seedling numbers present per plot

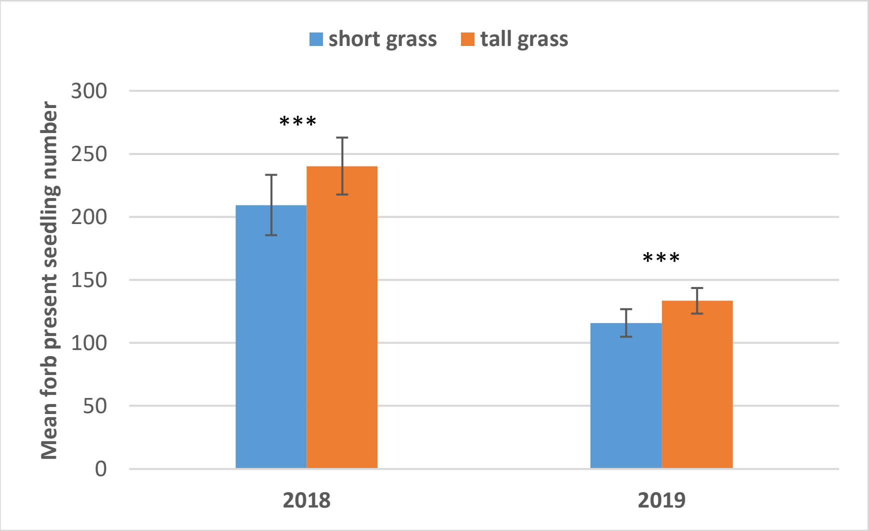


Figure 6.1 Effect of grass height on forb present seedling number/ plot in April 2018 and April 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

ii) Forb and grass biomass per plot

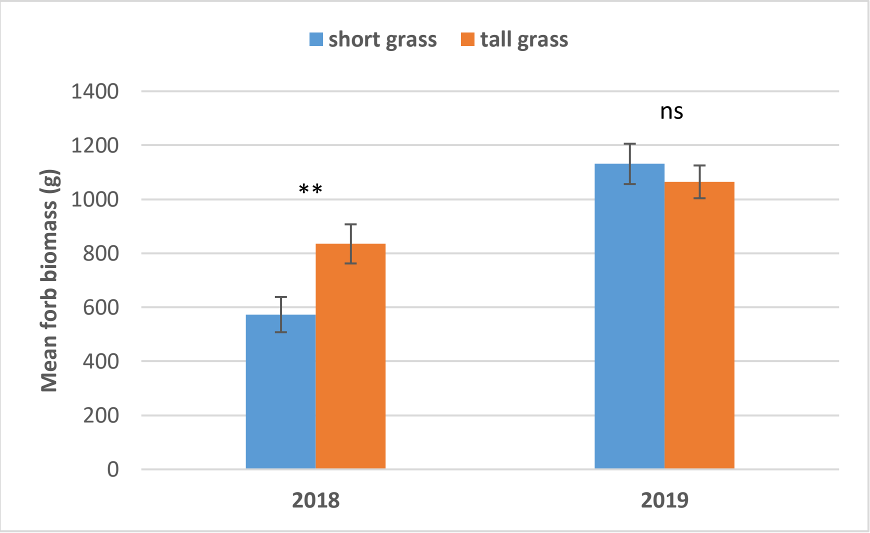


Figure 6.2 Effect of height of grass on forb biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.001 and ns=not significant. Error bar = 2 Standard Errors)

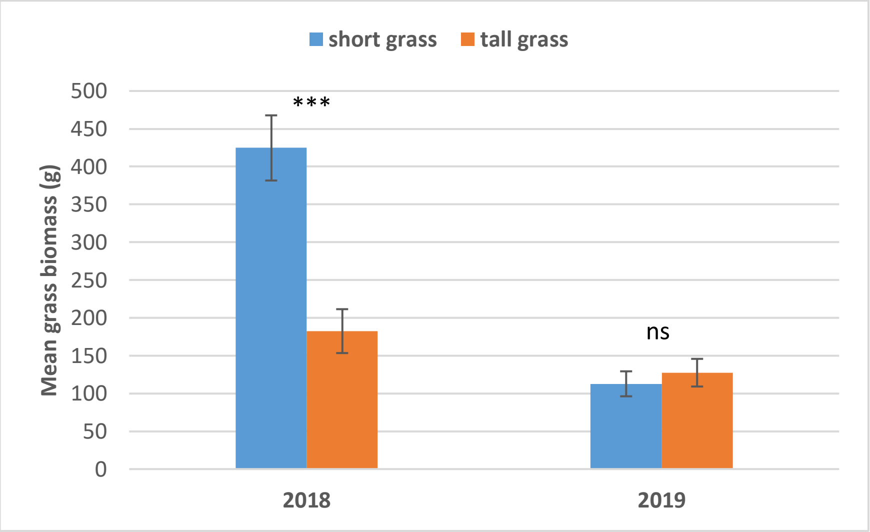


Figure 6.3 Effect of grass height on grass biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

iii) Forb, grass and bare ground cover values

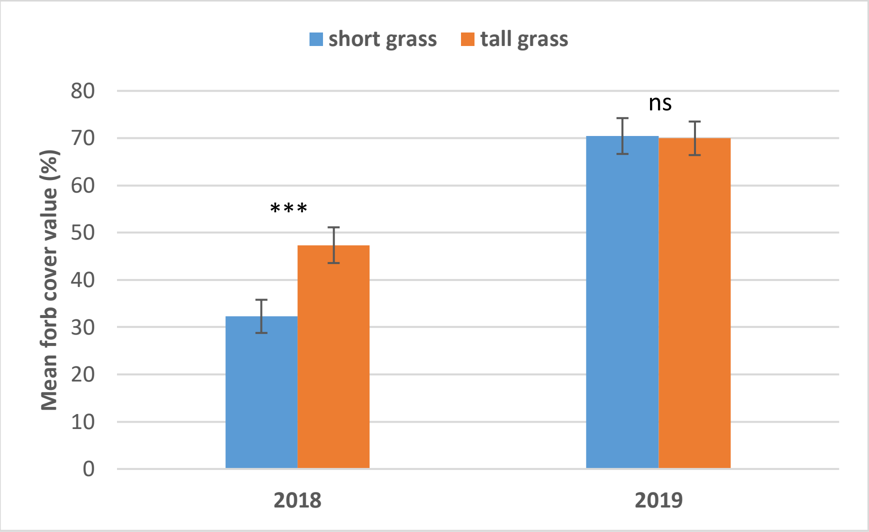


Figure 6.4 Effect of grass height on forb cover values in May 2018 and May 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

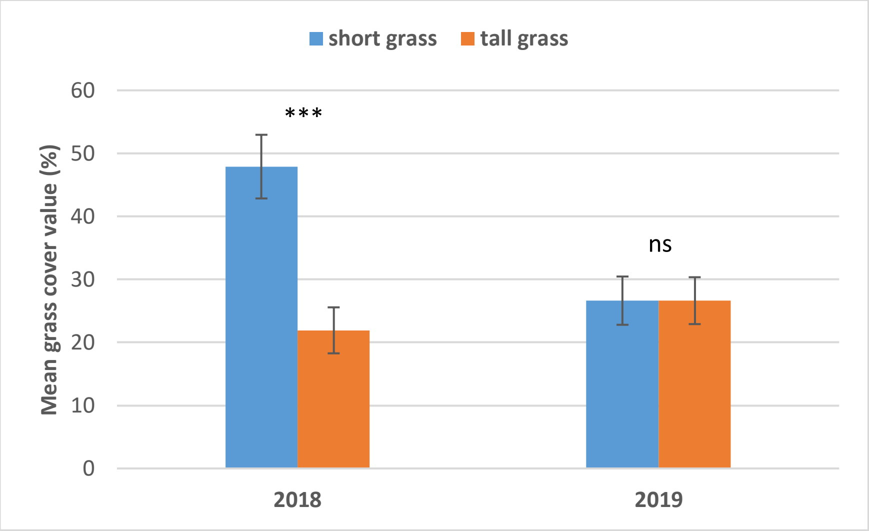


Figure 6.5 Effect of grass height on grass cover values in May 2018 and May 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

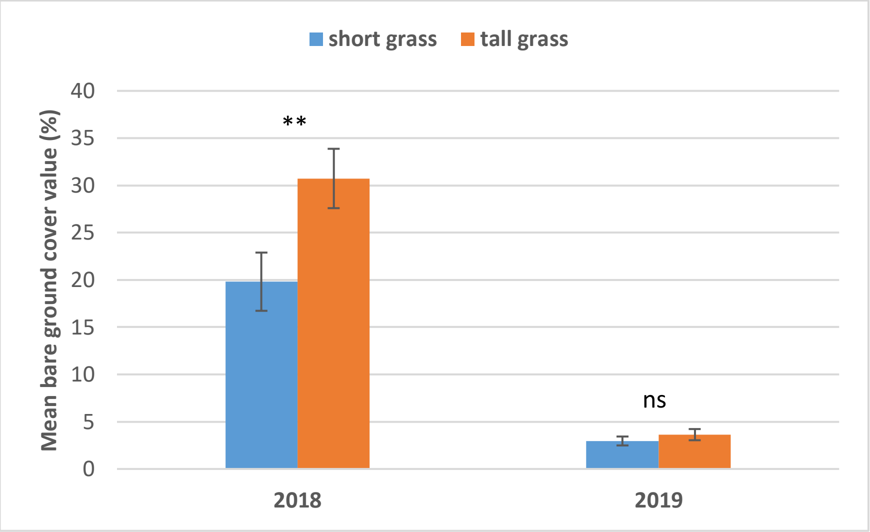


Figure 6.6 Effect of grass height bare ground cover values in May 2018 and May 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

6.2.2 Effect of grass height on dominant forbs (*Achillea millefolium* and *Echinops sphaerocephalus*) and subordinate forb biomass in 2018 and 2019

Achillea millefolium had significantly less biomass in the short grass treatment (425.83g) than the tall grass treatment (620.72g) in 2018 (p=0.004) (as shown in Figure 6.7). However, *A. millefolium* biomass significantly increased in the short grass treatment from 425.83g to 564.74g from 2018 to 2019 (p=0.000) while there was almost no change in the tall grass treatment in the same time period (from 620.72g in 2018 to 614.71g in 2019, p=0.853). This led to no significant difference in *A. millefolium* biomass between the two treatment in 2019 (p=0.324).

Despite *Echinops sphaerocephalus* having more biomass in the tall grass treatment (125.00g) than the short grass treatment (100.04g) in 2018, the difference was not statistically significant (p=0.204) (Figure 6.8). However, the tall grass treatment had significantly less *E. sphaerocephalus* biomass in 2019 (494.67g in the short grass treatment and 331.52g in the tall grass treatment, p=0.040).

The tall grasses appeared more impactful to *E. sphaerocephalus* than *A. millefolium* in 2019. However, subordinate forb biomass was significantly higher in the tall grass treatment in both 2018 and 2019 (p=0.000) (as shown in Figure 6.9).

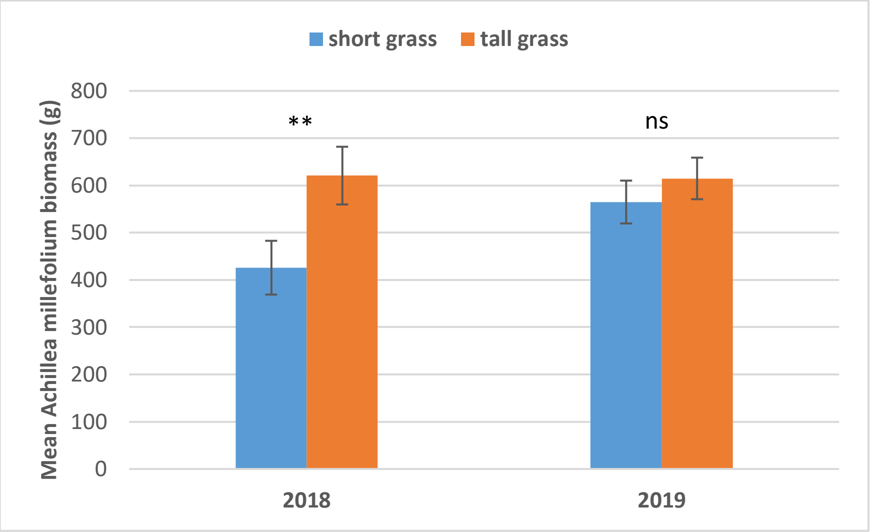


Figure 6.7 Effect of grass height on *Achillea millefolium* biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

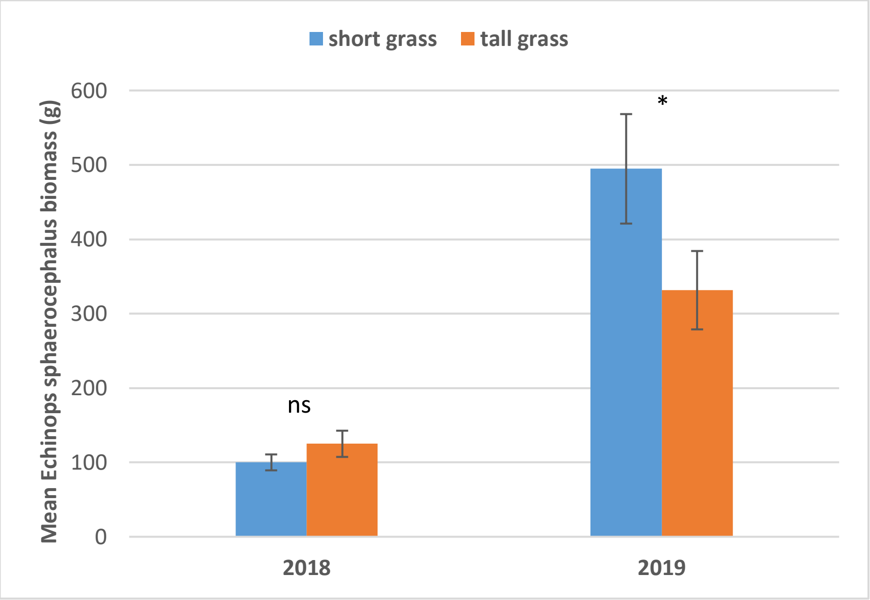


Figure 6.8 Effect of grass height on *Echinops sphaerocephalus* biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

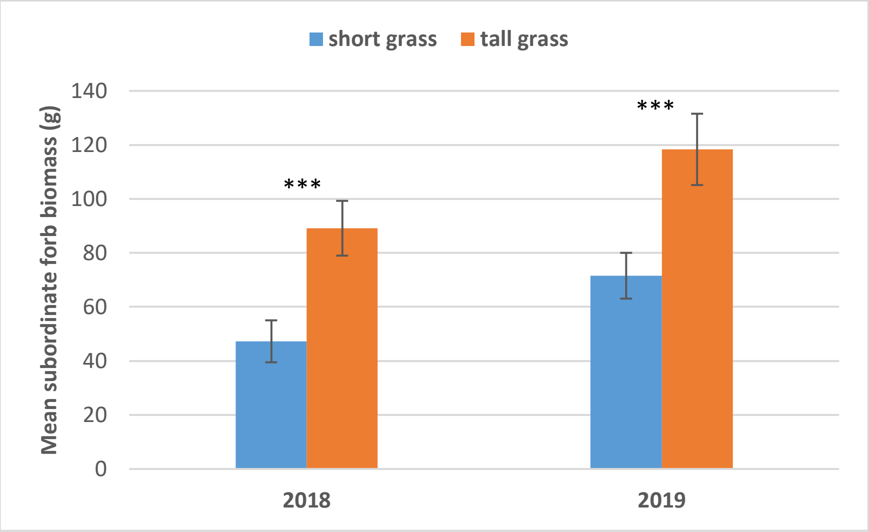


Figure 6.9 Effect of grass height on subordinate forb biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

Table 6.1 Effect of grass height on overall plot results in 2018 and 2019 (SE = Standard Error of Mean)

	2018					**2019**				
	Short grass		**Tall grass**		*P* value	**Short grass**		**Tall grass**		*P* value
	Mean	SE	Mean	SE		Mean	SE	Mean	SE	
Forb seedling present number	209.42	23.94	240.33	22.60	0.000 ***	115.83	10.95	133.40	10.16	0.000 ***
Forb biomass (g)	573.12	65.23	834.85	72.41	0.001 **	1130.96	74.68	1064.57	60.65	0.435 ns
Grass biomass (g)	424.69	43.11	182.44	29.02	0.000 ***	112.77	16.51	127.48	18.29	0.438 ns
Forb cover value (%)	32.29	3.51	47.35	3.78	0.000 ***	70.44	3.79	69.96	3.55	0.901 ns
Grass cover value (%)	47.90	5.05	21.92	3.65	0.000 ***	26.63	3.83	26.63	3.72	1.000 ns
Bare ground cover value (%)	19.81	3.08	30.73	3.14	0.002 **	2.96	0.47	3.63	0.59	0.331 ns
***Achillea millefolium* biomass (g)**	425.83	56.93	620.72	61.06	0.004 **	564.74	45.34	614.71	43.97	0.324 ns
***Echinops sphaerocephalus* biomass (g)**	100.04	10.71	125.00	17.63	0.204 ns	494.67	73.61	331.52	52.69	0.040 *
Subordinate forb biomass (g)	47.26	7.77	89.13	10.14	0.000 ***	71.55	8.48	118.33	13.20	0.000 ***

6.2.3 Effect of grass height on forb seedling number per species in 2018 and 2019

In 2018, there were nine forb species with significantly more seedlings in the tall grass treatment (refer to Table 6.2). The species included short canopy species of *Potentilla rupestris* (p=0.048), *Thermopsis lanceolata* (p=0.002), *Dracocephalum rupestre* (p=0.000) and *Dracocephalum ruychiana* (p=0.002); medium canopy species of *Campanula punctata* (p=0.000), *Delphinium grandiflorum* (p=0.007) and *Kalimeris incisa* (p=0.001); and tall canopy species of *Patrinia scabiosifolia* (p=0.000) and *Thalictrum aquilegifolium* (p=0.024).

In 2019, eleven forb species had significantly more seedlings in the tall grass treatment. Seven out of 11 of the low canopy forbs including *Anemone sylvestre* (p=0.001), *Potentilla rupestris* (p=0.001), *Thermopsis lanceolata* (p=0.004), *Dracocephalum rupestre* (p=0.000), *Dracocephalum ruychiana* (p=0.001), *Veronica teucrium* (p=0.016) and *Thalictrum petaloideum* (p=0.042). Medium canopy species of *Campanula punctata* (p=0.000), *Delphinium grandiflorum* (p=0.005) and *Kalimeris incisa* (p=0.000) had more seedling in the tall grass treatment. There was only one species; *Patrinia scabiosifolia* (p=0.019) that retained the similar pattern in 2019.

In both years, no forb species had significantly more seedlings in the short grass treatment.

Table 6.2 Effect of grass height on seedling number of each forb species in 2018 and 2019 (SE = Standard Error of Mean; na = not applicable)

	2018					**2019**				
	Short grass		**Tall grass**		*P* value	**Short grass**		**Tall grass**		*P* value
	Mean	SE	Mean	SE		Mean	SE	Mean	SE	
Low canopy										
Shared										
Anemone sylvestre	7.81	1.43	8.44	1.13	0.461 ns	4.21	0.80	5.71	0.74	0.001 **
Galium verum	9.04	1.24	9.56	1.62	0.254 ns	7.04	0.89	6.73	0.96	0.981 ns
Potentilla rupestris	9.48	1.26	9.90	1.02	0.048 *	6.08	0.71	7.88	0.75	0.001 **
Pulsatilla vulgaris	6.67	1.31	8.46	1.24	0.168 ns	2.21	0.60	3.15	0.56	0.056 ns
Veronica teucrium	8.27	1.38	9.10	1.21	0.058 ns	5.40	0.73	6.77	0.93	0.016 *
Mongolian										
Dracocephalum rupestre	14.50	2.00	18.38	1.94	0.000 ***	4.13	0.78	5.58	0.77	0.000 ***
Dracocephalum ruychiana	10.71	2.06	12.56	1.73	0.002 **	2.23	0.60	2.60	0.37	0.001 **
Thalictrum petaloideum	11.58	1.48	11.21	1.09	0.413 ns	5.79	0.76	6.13	0.75	0.042 *
Thermopsis lanceolata	7.38	1.43	9.40	1.24	0.002 **	1.42	0.33	2.38	0.39	0.004 **
Veronica incana	0.15	0.06	0.73	0.31	0.067 ns	0.00	0.00	0.00	0.00	0.154 ns
Medium canopy										
Shared										
Achillea millefolium	6.52	0.71	6.71	0.75	1.000 ns	6.52	0.71	6.71	0.75	1.000 ns
Origanum vulgare	25.85	2.70	28.35	3.14	0.084 ns	16.06	1.36	15.27	1.26	0.886 ns
Polemonium caeruleum	7.04	0.90	7.48	0.96	0.392 ns	3.02	0.46	3.92	0.84	0.224 ns
Stachys officinalis	6.67	0.74	6.67	0.74	na	4.08	0.52	4.19	0.52	0.168 ns
Mongolian										
Campanula punctata	19.44	2.97	27.06	2.69	0.000 ***	11.42	1.93	17.46	1.71	0.000 ***
Delphinium grandiflorum	12.52	1.95	13.29	1.60	0.007 **	7.42	1.23	9.60	1.36	0.005 **
Kalimeris incisa	9.04	1.34	11.67	1.38	0.001 **	4.42	0.70	6.21	0.81	0.000 ***
Platycodon grandiflorum	3.29	0.56	5.38	1.02	na	0.00	0.00	0.06	0.06	na
Scutellaria baicalensis	2.02	0.46	2.42	0.57	1.000 ns	0.21	0.07	0.17	0.07	na
Tall canopy										
Shared										
Echinops ritro	1.42	0.36	1.54	0.28	0.494 ns	0.83	0.25	0.73	0.19	0.781 ns
Geranium pratense	6.40	0.69	6.44	0.62	0.757 ns	7.31	0.87	6.08	0.57	0.088 ns
Sanguisorba officinalis	2.31	0.39	2.35	0.42	0.542 ns	2.02	0.28	1.98	0.30	0.543 ns
Thalictrum aquilegifolium	7.40	1.01	7.73	0.90	0.024 *	6.40	0.74	5.65	0.56	0.617 ns
Veronica longifolia	2.83	0.43	2.88	0.62	0.711 ns	1.98	0.36	2.71	0.55	0.234 ns
Mongolian										
Aconitum carmichaelii	1.00	0.00	1.00	0.00	na	0.85	0.05	0.81	0.06	0.552 ns
Angelica sylvestris	2.50	0.20	2.50	0.20	1.000 ns	0.60	0.11	0.65	0.11	0.979 ns
Echinops sphaerocephalus	4.83	0.38	4.77	0.38	0.819 ns	4.04	0.45	3.58	0.42	0.548 ns
Patrinia scabiosifolia	2.75	0.39	4.38	0.48	0.000 ***	0.15	0.05	0.71	0.33	0.019 *

6.2.4 Effect of grass height on subordinate forb biomass per species in 2018 and 2019

In both 2018 and 2019, the tall grass treatment more effectively increased the biomass of low and medium canopy forbs over tall canopy forbs.

As shown in Table 6.3, in 2018, the tall grass treatment significantly increased the biomass of subordinate forbs including low canopy forbs of *Anemone sylvestris* (p=0.019), *Potentilla rupestris* (p=0.010), *Pulsatilla vulgaris* (p=0.010), *Dracocephalum rupestre* (p=0.000), *Dracocephalum ruychiana* (p=0.000) and *Veronica teucrium* (p=0.002); medium canopy forbs of *Campanula punctata* (p=0.000), *Delphinium grandiflorum* (p=0.008), *Kalimeris incisa* (p=0.004) and *Platycodon grandiflorum* (p=0.000); and tall canopy forb of *Patrinia scabiosifolia* (p=0.000).

In 2019, low canopy subordinate forbs of *Galium verum* (p=0.015), *Thermopsis lanceolata* (p=0.007), *Veronica teucrium* (p=0.000) produced significantly more biomass in the tall grass treatment. Medium canopy subordinate forbs showed the same pattern were *Polemonium caeruleum* (p=0.017), *Campanula punctata* (p=0.008), *Delphinium grandiflorum* (p=0.041) and *Kalimeris incisa* (p=0.001). Tall forb species *Sanguisorba officinalis* (p=0.045) had significantly more biomass in the tall grass treatment.

Again, no subordinate forb had significantly more biomass in the short grass treatment in both years.

Table 6.3 Effect of grass height on biomass of each subordinate forb species in 2018 and 2019 (SE = Standard Error of Mean; na = not applicable)

	2018					**2019**				
	Short grass		**Tall grass**		*P* value	**Short grass**		**Tall grass**		*P* value
	Mean	SE	Mean	SE		Mean	SE	Mean	SE	
Low canopy										
Shared										
Anemone sylvestre	0.04	0.01	0.11	0.03	0.019 *	0.02	0.01	0.08	0.04	0.146 ns
Galium verum	2.01	0.69	4.02	1.22	0.080 ns	5.14	1.02	10.14	2.21	0.015 *
Potentilla rupestris	1.69	0.42	4.36	1.08	0.010 *	1.83	0.40	6.57	2.63	0.052 ns
Pulsatilla vulgaris	0.07	0.02	0.24	0.06	0.010 *	0.01	0.01	0.07	0.03	0.076 ns
Veronica teucrium	0.57	0.17	2.06	0.51	0.002 **	0.82	0.22	3.14	0.76	0.000 ***
Mongolian										
Dracocephalum rupestre	1.50	0.25	3.28	0.51	0.000 ***	0.08	0.03	0.11	0.03	0.461 ns
Dracocephalum ruychiana	0.42	0.10	1.31	0.23	0.000 ***	0.09	0.05	0.43	0.22	0.084 ns
Thalictrum petaloideum	0.00	0.00	0.00	0.00	na	0.00	0.00	0.00	0.00	0.110 ns
Thermopsis lanceolata	0.92	0.40	1.43	0.27	0.248 ns	0.13	0.05	0.77	0.25	0.007 **
Veronica incana	0.00	0.00	0.01	0.00	0.098 ns	0.00	0.00	0.00	0.00	na
Medium canopy										
Shared										
Origanum vulgare	14.71	3.29	21.27	3.38	0.085 ns	36.11	5.52	48.05	6.95	0.116 ns
Polemonium caeruleum	0.60	0.27	1.97	0.86	0.073 ns	0.49	0.19	1.95	0.65	0.017 *
Stachys officinalis	0.41	0.08	0.38	0.08	0.736 ns	1.34	0.49	0.99	0.21	0.468 ns
Mongolian										
Campanula punctata	0.61	0.22	3.23	0.73	0.000 ***	0.17	0.07	1.91	0.70	0.008 **
Delphinium grandiflorum	1.24	0.42	3.61	0.93	0.008 **	0.98	0.55	3.21	1.05	0.041 *
Kalimeris incisa	12.99	3.55	25.00	3.98	0.004 **	4.34	1.08	16.72	4.15	0.001 **
Platycodon grandiflorum	0.03	0.01	0.13	0.03	0.000 ***	0.01	0.00	0.01	0.01	0.320 ns

Scutellaria baicalensis	0.39	0.11	0.83	0.29	0.112 ns	0.04	0.02	0.14	0.06	0.062 ns
Tall canopy										
Shared										
Echinops ritro	2.92	0.91	4.63	1.50	0.303 ns	3.34	1.38	3.10	1.17	0.887 ns
Geranium pratense	3.10	0.82	4.04	0.63	0.291 ns	12.04	3.65	13.48	2.33	0.701 ns
Sanguisorba officinalis	1.72	0.53	3.65	0.95	0.059 ns	2.51	0.65	5.40	1.40	0.045 *
Thalictrum aquilegifolium	0.06	0.01	0.08	0.02	0.194 ns	0.03	0.01	0.06	0.03	0.203 ns
Veronica longifolia	0.72	0.40	2.29	0.79	0.053 ns	1.00	0.34	1.57	0.53	0.328 ns
Mongolian										
Aconitum carmichaelii	0.26	0.04	0.25	0.04	0.777 ns	0.42	0.09	0.23	0.07	0.065 ns
Angelica sylvestris	0.03	0.02	0.02	0.01	0.773 ns	0.57	0.52	0.07	0.03	0.297 ns
Patrinia scabiosifolia	0.23	0.07	0.92	0.20	0.000 ***	0.05	0.02	0.12	0.04	0.085 ns

6.3 Discussion

6.3.1 The effect of the grass height on overall numbers of forb seedling, biomass of forb and grass, and cover values

The original hypothesis was to test the response of the forb community to different heights of grasses, which play the role of competitive opponents. A pre-sowing germination test in the laboratory suggested an approximately 10% germination rate for *Deschampsia cespitosa* ‘Barchampsia’. This was factored into the sowing algorithm (i.e. more seed was added to compensate for low viability), but the resulting seedlings appeared to have much lower vigour than the shorter *Deschampsia cepitosa* selection. As a result, *Deschampsia cespitosa* ‘Barcampsia’, the “tall grass”, did not achieve a comparable seedling density or biomass to *Deschampsia cespitosa* ‘Pixie Fountain’ (the short grass) in the first two years (2017 and 2018). In terms of the cover values measured in early October 2017 the presence of the tall grass was significantly lower than the short grass (40% in the short grass treatment; 12% in the tall grass treatment, p=0.000). In spring 2018, the grass cover values increased to 48% in the short grass treatment and 22% in the tall grass treatment. This increase was probably because winter chilling stimulated more grass emergence from the initial sowing or from the additional sowing made in December 2017 (as explained in Chapter 3). However, the grass coverage presence between the two types was still different in 2018 (p=0.000). Thus, this discussion focuses more on competition generated by different grass seedling density and biomasses, rather than heights of the two *Deschampsia cespitosa* forms on the forb performance.

In 2018, the significantly higher grass biomass (p=0.000) in the short grass treatment led to a significant decrease of forb seedlings (p=0.001). These results agree with previous research on grass competition that forbs decline with an increase of grass density and or biomass (Sluis, 2002; Del-Val and Crawley, 2005; Dickson and Busby, 2009). Grass competition from the germination stage, can reduce forb seedling numbers, as the grasses are among the first species to emerge. It has been suggested that the grass canopy forms a physical barrier which can reduce amplitude of daily fluctuation in air temperature, which accompanies a reduction in the maximum daily air temperature (Bosy and Reader, 2006). Less fluctuation in air temperature can cause a low germination rate (Bannister, 1976). However, these findings were mostly derived from the lab environment. The results from the field experiment may be mainly ascribed to the seedling competition between the forb and grass. Grass seedlings with upright and denser tillers help to dominate light resource giving them superior survivorship to neighbouring forbs (del-Val and Crawley, 2005; Williams et al., 2007). This process restricts vegetative expansion of the forbs and may even eliminate individuals, which directly led to a reduction in forb seedlings and biomass.

To address the low biomass of the tall grasses, tall grass seedlings were transplanted into the experiment in June 2018 (as explained in Chapter 3). This was less effective in terms of boosting biomass of the tall grass by August 2018 than anticipated. By 2018, forbs had mostly closed the canopy and limited transplanted grasses access to light, although *Deschampsia cespitosa* is relatively shade tolerant for a grass.

There was a significant reduction from summer 2018 to 2019 in grass biomass, but the short grass experienced a greater loss (73.45% reduction in the short grass treatment and 30.12% reduction in the tall grass treatment, p=0.000 for both). This appears to be primarily due to the summer drought in 2018 and the spring in 2019, although it may also in part be due to increasing competition from the forb dominants. As discussed in Chapter 4, *Deschampsia cespitosa* are best adaptive to moist to wet soils. The aridity favoured the more stress tolerant forbs than the grasses. The sand mulch substrate had low water holding capacity which further limited the vigour for grasses and made forbs more competitive. By 2019, comparable biomasses of both tall and short *Deschampsia cespitosa* was achieved. The cover value of the both grasses were almost identical (27% for both, p=1.000) with a significant decrease in the short grass (p=0.000) and a significant increase in the tall grass (p=0.006) over the two years. The increase of the tall grasses was mostly ascribed to the transplanted seedlings.

The number of forb seedlings was still higher in the tall grass treatment in spring 2019 (p=0.000). This is presumably as a result of the higher forb elimination in the short grass treatment in 2018 plus that surviving forbs more established in the tall grass treatment before grass biomass became similar. However, the higher seedling number did not lead to a difference in coverage (approximately 70% in both the short and tall grass treatment, p=0.901). In summer, the biomass of two grass types were similar (p=0.438) and there were no significant effects of grass type on the production of forb biomass (p=0.435). This suggests that the difference in grass heights did not have a major impact overall on an established forb community in the third year. Clearly there is an element of catch up operating within the data in year three. Although the grass biomass has become broadly equivalent between the two *Deschampsia cespitosa*, the forbs in the taller form had previously benefitted from lesser competition. It would have been interesting to see if forb biomass would have begun to respond as originally hypothesised in the fourth year. The tall grasses were observed to grow faster in spring and early summer in 2019. This suggests that the height advantage of ‘Barcampsia’ would make it more competitive to forbs in the future.

6.3.2 The effect of the grass height on the biomass of dominant forbs (*Achillea millefolium* and *Echinops sphaerocephalus*) and subordinate forbs

In 2018, the higher amount of grass presence in the short grass treatment led to a significant reduction on the biomass of *Achillea millefolium* (p=0.004) and the subordinates (p=0.000). The presence of grasses from the seedling stage restricted the rhizomatous expansion of *A. millefolium* and the light access to the subordinate forbs. Grass competition caused biomass reduction of both dominant and subordinate forbs. However, although *E. sphaerocephalus* developed 25% more biomass in the tall grass treatment, overall grasses had less effect on the growth of *E. sphaerocephalus* (p=0.204) than on *A. millefolium*. The growth rate of *E. sphaerocephalus*, capacity for “shoot thrust” and shoot height allowed it to overcome the ‘grass barrier’ (Figure 6.10).

By 2019, the grass treatments no longer had significant effects on the biomass of *A. millefolium* (p=0.324), this species effectively escaped the grass competition, by virtue of its tall leafy stems. The biomass of the *E. sphaerocephalus* was however reduced in the tall grass treatment, suggesting that the taller canopy was perhaps interfering with the photosynthesis of its predominantly basal leaves (refer to Figure 6.11).

The decline of the short grass biomass led to an increase in the biomass of dominant forbs in 2019. Both *A. millefolium* and *E. sphaerocephalus* took the ‘grassy dominant role’ and kept imposing the competitive stress on the subordinate forbs in the short grass treatment in 2019, despite the total biomass of subordinate forbs not increasing in both grass treatments from 2018 to 2019. Besides, the most competitive subordinates may also have contributed to suppressing the growth of *E. sphaerocephalus* in the tall grass treatment in spring 2019.

In general, the subordinate forbs did not benefit greatly from the reduction of grass biomass as the major competition shifted from being between subordinate forbs and grasses to being between the dominant forbs and subordinate forbs. This suggests that grass removal proposed in a management plan may cause the development of forb dominants rather than increasing the subordinate forb biomass.

Figure 6.10 *Deschampsia cespitosa* ‘Pixie Fountain’ had limited constraints on the growth of *Echinops sphaerocephalus* in both spring and summer (Picture on the left taken on the 5th May and the right one taken on the 26th July 2018)

Figure 6.11 The L2 and H3 (tall grass treatment, top left and right) had much lower *Echinops sphaerocephalus* biomass than the H3 and M3 plots (short grass treatment, bottom left and right) in late May 2019. The difference became more evident in the plots where had high grass density (high sowing density x forb: grass sowing ratio of 1: 9) in 75mm sand substrate. Tall grasses intercepted more lights than the short grass from spring. *Echinops sphaerocephalus* had predominantly basal leaves and the biomass production were sensitive to the difference in grass heights in 2019.

6.3.3 The effect of grass height on the number of seedlings and biomass of subordinate forb species

The general pattern was that no subordinate species had significantly greater seedling numbers or biomass in the short grass treatment in 2018. This suggests that grass competition was more effective than the dominant forb competition on both subordinate species seedling numbers and biomass. This pattern was repeated in 2019 despite the decline of the short grass, indicating that vegetative expansion of the dominant forbs in the short grass treatment preserved the suppression on the subordinates. *Echinops sphaerocephalus* developed significant more biomass in this treatment.

In 2018, subordinate forbs in the short grass treatment suffered from more grass competition while they were more affected by *Achillea millefolium* competition in the tall grass treatment. Grass had denser shoots from the seedling stage and was more likely to restrict light access to the subordinate, than the spatially distanced *A. millefolium*.

The species which had the greatest reduction in seedling number and biomass in the short grass treatment in 2018 were those which had the lowest capacity to push their shoots into the light through the grass foliage. In terms of seedling number, 9 out of 26 subordinate species had significantly greater differences between the short grass and tall grass treatment, with *Dracocephalum rupestre*, *Campanula punctata* and *Patrinia scabiosifolia* the species with the greatest differences (p=0.000). These are coincidently Inner Mongolian species and whilst reduced climatic fitness may have played a role, it seems likely this was more about the morphology of the seedlings. *D. rupestre* and *C. punctata* have typically low horizontal orientation of cotyledons, which make them too difficult to push upwards though the grass tillers (Bosy and Reader, 2006). *P. scabiosifolia* is a late emerging species, making it more difficult for the seedlings to break through the grass canopy when it was becoming denser in late spring (Hitchmough, 2009). In terms of subordinate biomass there is a disproportionate pattern of the effects on the species on each canopy layer. Six of the species in the lowest canopy layer were significantly higher in biomass in the tall grass treatment (i.e. associated with low grass biomass), with four species in the intermediate layer and only one species in the tallest layer. This suggests that greater canopy height of forbs may increase their competitiveness under grass competition (Vojtech, Turnbull and Hector, 2007).

The species whose biomass was the least sensitive to the treatment appeared to be large seeded species such as *Thermopsis lanceolata* (p=0.248), *Geranium pratense* (p=0.701) and *Echinops ritro* (p=0.303) (as well as *Stachys officinalis*, a transplanted species as explained in Chapter 5, p=0.736). Those species were likely well established at the seedling stage. Their larger seed size in the mix suggests larger carbohydrate reserves which may increase the seedling survival and potentially growth rate (Jurado and Westoby, 2006; Ben-Hur and Kadmon, 2015). Seedlings that are taller at the emergence stage can make the species more competitive in later stages (Weiner, 1990; Tremmel and Bazzaz, 1993). Large seeded species have more capacity to thrust through dense productive vegetation, useful trait in designed communities (Bjørn *et al.*, 2019). Moreover, architectural advantages in light competition such as *G. pratense* long petiole with the leaf on the top and *E. ritro* 'shoot thrust' ability (Grime, 2002) also facilitate persistence. Species with traits that are associated with superior light competition such as early emergence (e.g. *Origanum vulgare*, p=0.085) and tall leafy stems (e.g. *Veronica longifolia*, p=0.053) or relatively high shade tolerance (e.g. *Polemonium caeruleum*, p=0.073) showed no statistical difference between the two treatments.

In 2019, the level of light availability for subordinate forbs became similar in June and July. The Sunscan PAR measures indicate almost identical solar levels on the soil surface (2.40% in the short grass and 2.35% in the tall grass in June 2019, p=0.914; 3.96% in the short grass and 3.96% in the tall grass in July 2019, p=0.998). Also, the grass coverage was similar in spring. However, this did not level out the number of seedlings of species between the two grass treatments (i.e. the effects of the denser grass in the short treatment were still evident). This suggests the seedling emergence and survivorship in the first one or two years are essential for long-term establishment in the community (Lauenroth and Adler, 2008), despite seedlings remaining small. The impact of the dense grass biomass in year one on these species, was still having a negative effect even though that biomass was now in decline, albeit replaced by the increasing biomass dominance of *Achillea millefolium*. Maximising seedling survival is commonly the priority for the practitioners when trying to establish a sown meadow community (Hitchmough, 2017a) as it opens up at least the possibility of future expansion and colonisation. What is more, a few more small species appeared to have significantly fewer seedlings in the short grass treatment, e.g. *Anemone sylvestris* (p=0.001), *Veronica teucrium* (p=0.016) and *Thalictrum petaloideum* (p=0.042). This was ascribed to the continuous grass competition in summer 2018 and again suggests the lowest canopy species were more sensitive to grass biomass. In terms of biomass, more forb species within the low canopy group appeared to have no significant difference in the treatments. This suggests those species were suppressed in both treatments as *Achillea millefolium* took the ‘grassy role’ and the lowest canopy forbs were sensitive to this. The species showed no significant differences in 2019 between the short and tall grass treatment but persisted with substantial biomass in both treatments are likely to be the ones that can persist in these conditions. Those species were *Origanum vulgare* (p=0.116), *Geranium pratense* (p=0.701) and *Echinops ritro* (p=0.887) etc. Abilities to compete for light including upright growth form, rapid seedling growth in spring and rapid recovery after summer cutback etc, is the key to long term persistence in the community (refer to Figure 6.12 and 6.13).

Despite native species tending to be less sensitive to the density of grass seedlings, light competitiveness at the seedling stage is the key to survive and produce more biomass under grass competition. Also, the low canopy forbs generally appeared more sensitive to grass competition that indicates the greater risk of elimination or low biomass. This agrees with the suggestion that base layer species need to be significantly shade tolerant if they are not to be outcompeted by the dominants or emerging weeds (Hitchmough, 2017b; Hitchmough, Wagner and Ahmad, 2017). In the longer term, advantages in seedling architecture to access more light is essential for subordinates to persist with dominants. No subordinate reversed the pattern to develop more biomass in the short grass treatment in 2019 indicate that the subordinate species was not as sensitive to different heights of grass but rather the overall community dominants.

Figure 6.12 (left) *Achillea millefolium* have a faster recovery than *Deschampsia cespitosa* post cutting. They also emerge earlier in spring than grasses. In addition, *Achillea millefolium* are rhizomatous which have morphological competitive advantages (taken on 5th Sept 2018, few weeks after cutback)

Figure 6.13 (right) shows the evidence that *Echinops sphaerocephlus* suppresses surrounding grasses with their basal foliage rosette. The rapid recovery of *Geranium pratense*, and *Origanum vulgare* were also evident (taken on 25th Sept 2019, few weeks after cutback)

Chapter 7 Effects on depths of sand substrate

7.1 Introduction

Sands have been shown to be an effective substrate to limit the growth of vigorous species and weeds invasion but can still support a good emergence for desired species (Hitchmough & Fleur, 2006; Hitchmough & Wagner, 2013). It also gives sufficient soil moisture capacity and allows root penetration for the later establishment (Hitchmough et al., 2001). A depth of 75mm for sand mulch was suggested in Hitchmough & Wagner (2013) as the depth facilitates high emergence and establishment of sown species whilst preventing the emergence of most weedy species from the underlying soil.

This study looked at the effect of 75mm and 150mm depths of sand substrates on the grassland communities, especially the competition pattern between the forbs and grasses. By distancing from the fertile soil beneath, the deeper sand substrate decreased water availability at the surface of the substrate. The availability of water may result in a shift in the pattern of survivorship more than nutrients in a community (Grime, 2002). Also, the deeper sand layer can delay plant roots to contact with the fertile soil beneath. This study aimed to test if the deeper sand mulch treatment by significantly increasing the moisture stress to the community, can potentially decrease grass competition and enhance forb diversity and richness. In addition, can moisture stress overcome dominant effects and increase subordinate performance?

7.2 Results

7.2.1 Effect of depth of sand substrate on forb seedling numbers, forb and grass biomass and cover values in 2018 and 2019

As shown in Figure 7.1, the deeper sand substrate treatment (150mm) significantly increased the numbers of forb seedlings in both 2018 and 2019 (p=0.000). There was no significant difference in forb biomass however between two depths of sand mulch layer in 2018 (p=0.670) although the 150mm (721.52g) had slightly more biomass than the 75mm treatment (686.45g). In 2019, the 150mm treatment significantly increased forb biomass (1196.55g) compared to the 75mm treatment (998.99g), p=0.020) (Figure 7.2).

The 75mm treatment supported significantly more grass biomass (374.49g) than the 150mm treatment (232.63g, p=0.001) in 2018. In 2019, despite the 75mm still supporting more grass biomass (129.85g) than the 150mm treatment (110.40g), the difference was non-significant (p=0.306) (Figure 7.3).

Forb cover values showed similar trends to the results of forb biomass. There was no significant difference in 2018 (p=0.918) but the 150mm significantly increased forb cover value in spring 2019 (p=0.017) (Figure 7.4). However, the shallower sand substrate (75mm) plots significantly increased grass coverage in spring (p=0.000 in 2018 and p=0.006 in 2019) and significantly reduced bare ground coverage in both years (p=0.000 in 2018 and p=0.025 in 2019) (as shown in Figure 7.5 and 7.6).

i) Forb seedling numbers present per plot

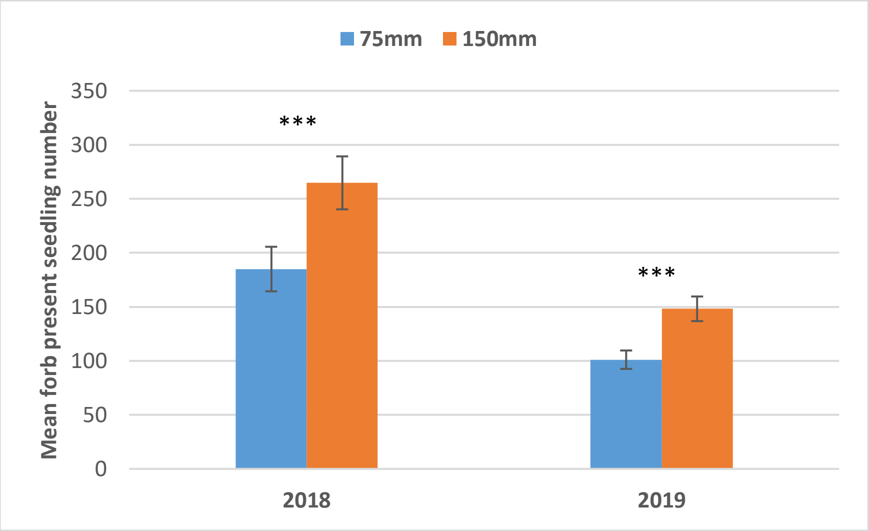


Figure 7.1 Effect of depth of sand substrate on forb present seedling number/ plot in April 2018 and April 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

ii) Forb and grass biomass per plot

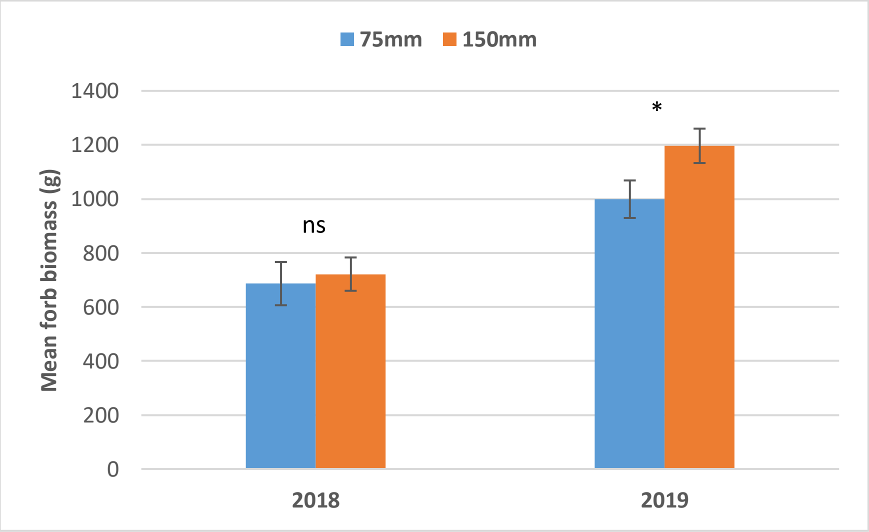


Figure 7.2 Effect of depth of sand substrate on forb biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.001 and ns=not significant. Error bar = 2 Standard Errors)

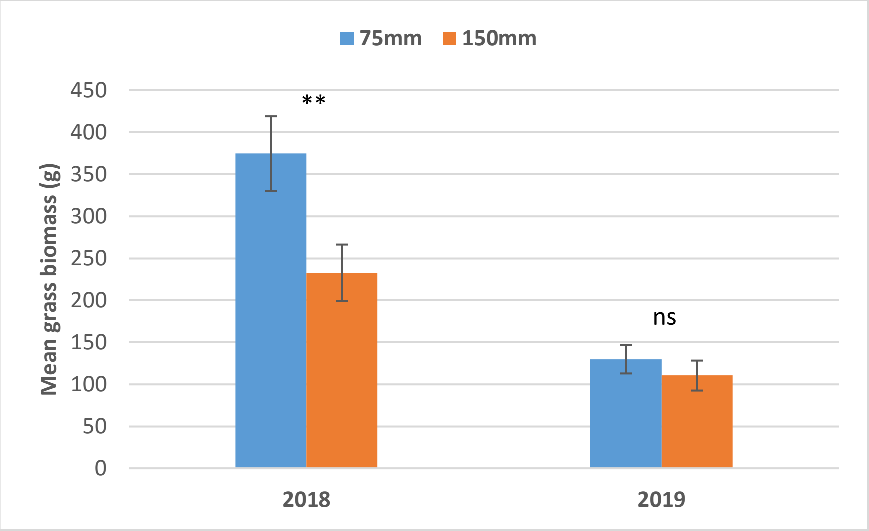


Figure 7.3 Effect of depth of sand substrate on grass biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

iii) Forb, grass and bare ground cover values

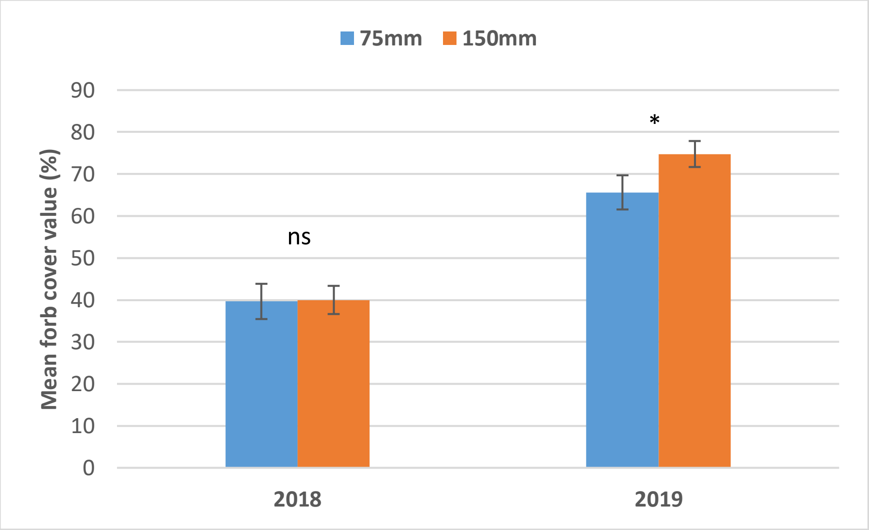


Figure 7.4 Effect of depth of sand substrate on forb cover values in May 2018 and May 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

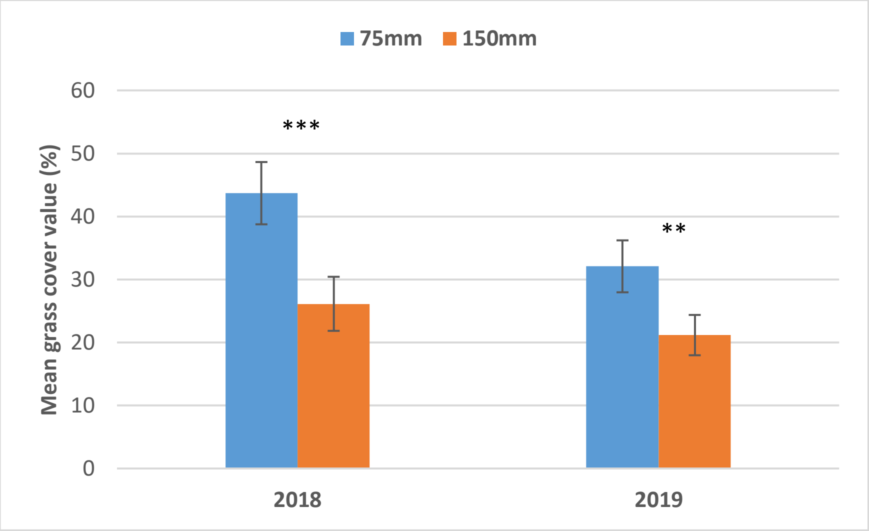


Figure 7.5 Effect of depth of sand substrate on grass cover values in May 2018 and May 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

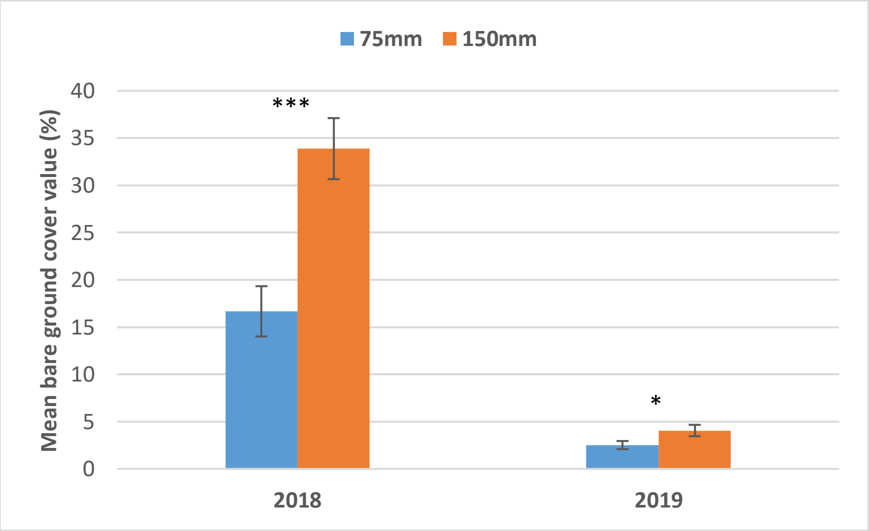


Figure 7.6 Effect of depth of sand substrate on bare ground cover values in May 2018 and May 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

7.2.2 Effect of depth of sand substrate on dominant forbs (*Achillea millefolium* and *Echinops sphaerocephalus*) and subordinate forb biomass in 2018 and 2019

As shown in Figure 7.7, *Achillea millefolium* biomass had no significant difference between the 75mm (495.96g in 2018 and 568.42g in 2019) and the 150mm treatment (550.59g in 2018 and 611.03g in 2019) in both years (p=0.416 in 2018 and p=0.400 in 2019), despite that 150mm supported slightly more biomass of *A. millefolium.*

The 150mm treatment supported more biomass of *Echinops sphaerocephalus* in both years that the 75mm treatment had 97.13g in 2018 and 308.50g in 2019 whereas the 150mm treatment had 308.50g in 2018 and 517.70g in 2019. These differences were not statistically significant in 2018 (p=0.117) but became significant in 2019 (p=0.008) (Figure 7.8).

However, the 75mm treatment significantly increased subordinate forb biomass in both years (p=0.000). This treatment almost doubled the subordinate biomass (2.2 times in 2018 and 1.8 times in 2019) (as shown in Figure 7.9).

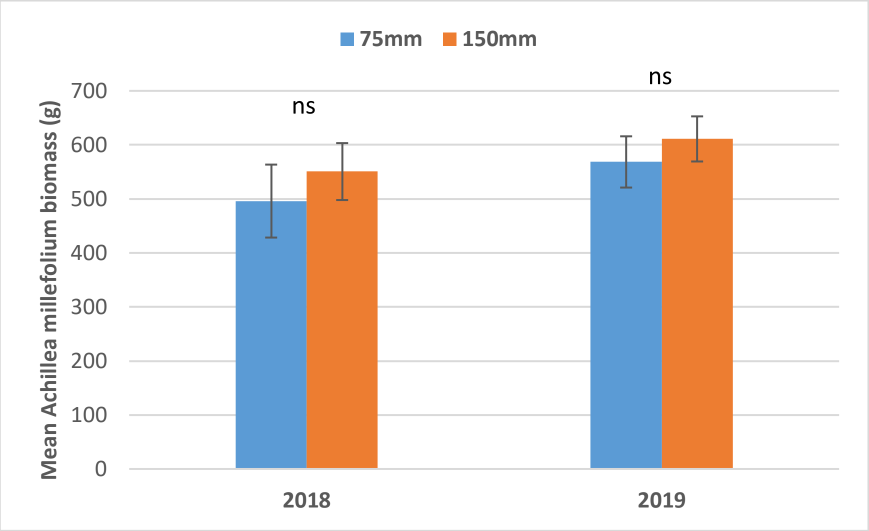


Figure 7.7 Effect of depth of sand substrate on *Achillea millefolium* biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

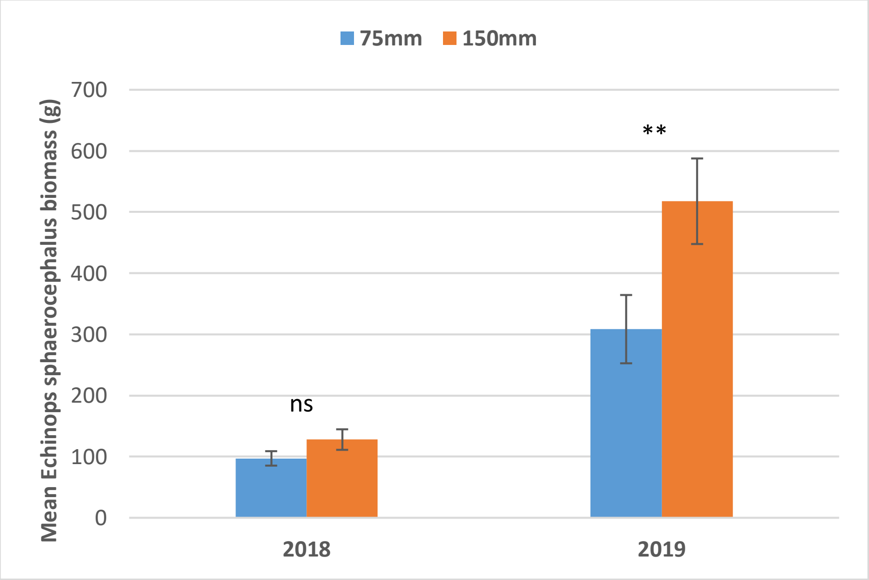


Figure 7.8 Effect of depth of sand substrate on *Echinops sphaerocephalus* biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

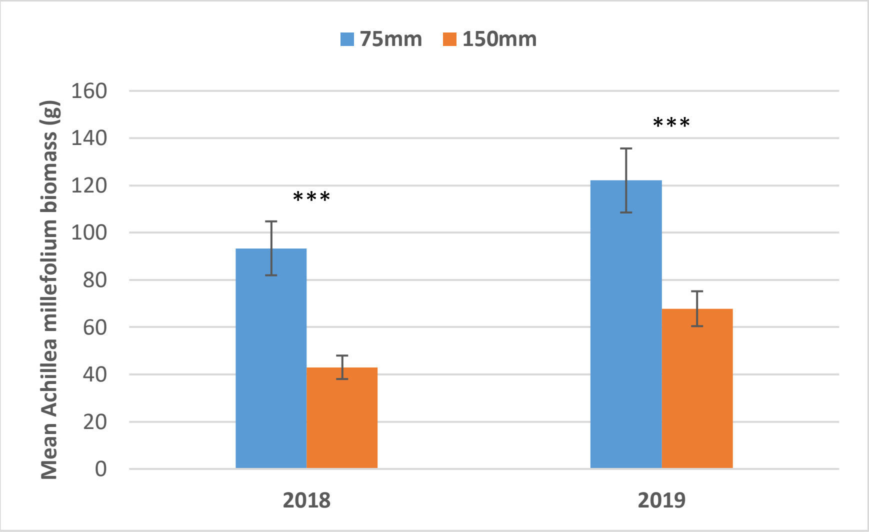


Figure 7.9 Effect of depth of sand substrate on subordinate forb biomass/ plot in August 2018 and August 2019 (*p ≤ 0.05; **p ≤ 0.01; ***p ≤ 0.000 and ns=not significant. Error bar = 2 Standard Errors)

Table 7.1 Effect of depth of sand substrate on overall plot results in 2018 and 2019 (SE = Standard Error of Mean)

	2018					**2019**				
	75mm		**150mm**		*P* value	**75mm**		**150mm**		*P* value
	Mean	SE	Mean	SE		Mean	SE	Mean	SE	
Forb seedling present number	184.98	20.62	264.77	24.52	0.000 ***	101.06	8.52	148.17	11.41	0.000 ***
Forb biomass (g)	686.45	79.98	721.52	61.79	0.670 ns	998.99	69.54	1196.55	63.65	0.020 *
Grass biomass (g)	374.49	44.46	232.63	33.68	0.001 **	129.85	16.95	110.40	17.83	0.306 ns
Forb cover value (%)	39.65	4.20	40.00	3.36	0.918 ns	65.63	4.07	74.77	3.10	0.017 *
Grass cover value (%)	43.69	4.94	26.13	4.29	0.000 ***	32.08	4.12	21.17	3.20	0.006 **
Bare ground cover value (%)	16.67	2.66	33.88	3.23	0.000 ***	2.52	0.43	4.06	0.60	0.025 *
***Achillea millefolium* biomass (g)**	495.96	67.54	550.59	52.72	0.416 ns	568.42	47.44	611.03	41.78	0.400 ns
***Echinops sphaerocephalus* biomass (g)**	97.13	11.82	127.91	16.81	0.117 ns	308.50	55.84	517.70	69.96	0.008 **
Subordinate forb biomass (g)	93.37	11.41	43.02	4.96	0.000 ***	122.08	13.55	67.81	7.39	0.000 ***

7.2.3 Effect of depth of sand substrate on forb seedling number per species in 2018 and 2019

The 150mm treatment significantly increased the number of seedlings for 12 forb species in 2018 (as in Table 7.2). This included 7 low canopy forbs of *Anemone sylvestris* (p=0.000), *Pulsatilla vulgaris* (p=0.011), *Thermopsis lanceolata* (p=0.000), *Dracocephalum rupestre* (p=0.000), *Dracocephalum ruychiana* (p=0.000), *Veronica teucrium* (p=0.020) and *Thalictrum petaloideum* (p=0.000). Within the medium canopy group, the species that had significantly positive response to the 150mm treatment were *Polemonium caeruleum* (p=0.000), *Campanula punctata* (p=0.000) and *Delphinium grandiflorum* (p=0.000). The tall canopy forbs with the similar pattern were *Patrinia scabiosifolia* (p=0.020) and *Thalictrum aquilegifolium* (p=0.003).

In 2019, this pattern was retained; low canopy forbs appeared more positively affected by the 150mm treatment. *Anemone sylvestre* (p=0.000), *Pulsatilla vulgaris* (p=0.000), *Dracocephalum rupestre* (p=0.000), *Dracocephalum ruychiana* (p=0.002), *Veronica teucrium* (p=0.001) and *Thalictrum petaloideum* (p=0.000) still had significantly more numbers of seedling in the 150mm treatment. *Campanula punctata* (p=0.000), *Delphinium grandiflorum* (p=0.000) and *Thalictrum aquilegifolium* (p=0.006) also retained the same pattern.

In both years, no forb species had significantly a greater number of seedlings in the 75mm treatment.

Table 7.2 Effect of depth of sand substrate on seedling number of each forb species in 2018 and 2019 (SE = Standard Error of Mean; na = not applicable)

	2018					**2019**				
	75mm		**150mm**		*P* value	**75mm**		**150mm**		*P* value
	Mean	SE	Mean	SE		Mean	SE	Mean	SE	
Low canopy										
Shared										
Anemone sylvestris	6.17	1.25	10.08	1.26	0.000 ***	2.56	0.43	7.35	0.89	0.000 ***
Galium verum	8.56	1.16	10.04	1.67	0.407 ns	6.81	0.82	6.96	1.02	0.861 ns
Potentilla rupestris	9.06	1.12	10.31	1.16	0.229 ns	6.52	0.73	7.44	0.74	0.188 ns
Pulsatilla vulgaris	6.25	1.36	8.88	1.17	0.011 *	1.19	0.34	4.17	0.69	0.000 ***
Veronica teucrium	7.04	1.04	10.33	1.47	0.020 *	4.38	0.45	7.79	1.05	0.001 **
Mongolian										
Dracocephalum rupestre	12.69	1.84	20.19	1.99	0.000 ***	3.29	0.61	6.42	0.87	0.000 ***
Dracocephalum ruychiana	8.52	1.58	14.75	2.08	0.000 ***	1.29	0.28	3.54	0.61	0.002 **
Thalictrum petaloideum	8.88	1.19	13.92	1.30	0.000 ***	3.77	0.51	8.15	0.82	0.000 ***
Thermopsis lanceolata	6.00	1.08	10.77	1.49	0.000 ***	2.08	0.43	1.71	0.30	0.658 ns
Veronica incana	0.44	0.27	0.44	0.17	0.988 ns	0.00	0.00	0.00	0.00	0.473 ns
Medium canopy										
Shared										
Achillea millefolium	6.50	0.72	6.73	0.74	0.241 ns	6.50	0.72	6.73	0.74	0.241 ns
Origanum vulgare	25.42	2.44	28.79	3.34	0.213 ns	15.06	1.15	16.27	1.44	0.455 ns
Polemonium caeruleum	5.65	0.76	8.88	1.03	0.000 ***	2.75	0.42	4.19	0.85	0.117 ns
Stachys officinalis	6.67	0.74	6.67	0.74	na	3.94	0.51	4.33	0.52	0.190 ns
Mongolian										
Campanula punctata	14.25	2.18	32.25	2.91	0.000 ***	8.10	1.33	20.77	1.89	0.000 ***
Delphinium grandiflorum	7.19	1.11	18.63	1.94	0.000 ***	4.06	0.63	12.96	1.47	0.000 ***
Kalimeris incisa	11.15	1.59	9.56	1.11	0.866 ns	5.88	0.86	4.75	0.66	0.570 ns
Platycodon grandiflorum	2.50	0.45	6.17	1.02	na	0.06	0.06	0.00	0.00	na
Scutellaria baicalensis	0.98	0.19	3.46	0.66	na	0.06	0.04	0.31	0.09	na
Tall canopy										
Shared										
Echinops ritro	1.27	0.29	1.69	0.35	0.165 ns	0.75	0.23	0.81	0.22	0.510 ns
Geranium pratense	6.31	0.68	6.52	0.64	0.141 ns	6.25	0.65	7.15	0.82	0.458 ns
Sanguisorba officinalis	2.58	0.42	2.08	0.38	0.438 ns	2.02	0.31	1.98	0.27	0.612 ns
Thalictrum aquilegifolium	6.69	0.96	8.44	0.93	0.003 **	5.58	0.71	6.46	0.59	0.006 **
Veronica longifolia	2.88	0.58	2.83	0.49	0.945 ns	2.56	0.47	2.13	0.46	0.751 ns
Mongolian										
Aconitum carmichaelii	1.00	0.00	1.00	0.00	na	0.83	0.05	0.83	0.05	0.946 ns
Angelica sylvestris	2.50	0.20	2.50	0.20	1.000 ns	0.65	0.10	0.60	0.11	0.782 ns

Echinops sphaerocephalus	4.58	0.33	5.02	0.42	0.353 ns	3.44	0.44	4.19	0.42	0.139 ns
Patrinia scabiosifolia	3.27	0.43	3.85	0.47	0.020 *	0.67	0.33	0.19	0.06	0.099 ns

7.2.4 Effect of depth of sand substrate on subordinate forb biomass per species in 2018 and 2019

According to Table 7.3, *Galium verum* (p=0.002 in 2018, p=0.001 in 2019), *Dracocephalum ruychiana* (p=0.023 in 2018, p=0.024 in 2019), *Veronica teucrium* (p=0.030 in 2018, p=0.010 in 2019), *Origanum vulgare* (p=0.000 in both years) and *Geranium pratense* (p=0.012 in 2018, p=0.029 in 2019) all had significantly more biomass in the 75mm treatment in both years.

The 75mm treatment significantly increased the biomass of *Potentilla rupestris* (p=0.007), *Kalimeris incisa* (p=0.000), *Thalictrum aquilegifolium* (p=0.034) and *Patrinia scabiosifolia* (p=0.050) in 2018 but not in 2019. However, *Polemonium caeruleum* had more biomass in the 75mm treatment in 2019 (p=0.045) but no significant difference in 2018 between the treatments.

Opposite to the results for forb seedling numbers, biomass of no subordinate species significantly increased in the 150mm treatment with the exemption of *Delphinium grandiflorum* in 2018 (1.54g in the 75mm treatment, 3.31g in the 150mm treatment, p=0.048).

Table 7.3 Effect of depth of sand substrate on biomass of each subordinate forb species in 2018 and 2019 (SE = Standard Error of Mean; na = not applicable)

	2018					2019				
	75mm		150mm		*P* value	75mm		150mm		*P* value
	Mean	SE	Mean	SE		Mean	SE	Mean	SE	
Low canopy										
Shared										
Anemone sylvestre	0.09	0.03	0.06	0.02	0.383 ns	0.07	0.04	0.03	0.01	0.379 ns
Galium verum	4.79	1.33	1.24	0.30	0.002 **	11.09	2.23	4.19	0.86	0.001 **
Potentilla rupestris	4.42	1.11	1.63	0.32	0.007 **	6.32	2.65	2.07	0.36	0.081 ns
Pulsatilla vulgaris	0.18	0.06	0.13	0.03	0.373 ns	0.05	0.03	0.03	0.01	0.646 ns
Veronica teucrium	1.84	0.50	0.79	0.23	0.030 *	2.83	0.75	1.13	0.29	0.010 *
Mongolian										
Dracocephalum rupestre	2.37	0.47	2.41	0.36	0.929 ns	0.06	0.03	0.13	0.04	0.079 ns
Dracocephalum ruychiana	1.12	0.24	0.61	0.10	0.023 *	0.49	0.22	0.03	0.01	0.024 *
Thalictrum petaloideum	0.00	0.00	0.00	0.00	na	0.00	0.00	0.00	0.00	0.110 ns
Thermopsis lanceolata	1.41	0.31	0.94	0.37	0.290 ns	0.61	0.23	0.30	0.12	0.200 ns
Veronica incana	0.00	0.00	0.00	0.00	0.297 ns	0.00	0.00	0.00	0.00	1.000 ns
Medium canopy										
Shared										
Origanum vulgare	26.26	4.27	9.73	1.26	0.000 ***	55.65	7.53	28.52	3.95	0.000 ***
Polemonium caeruleum	1.95	0.86	0.61	0.27	0.078 ns	1.83	0.65	0.61	0.20	0.045 *
Stachys officinalis	0.40	0.08	0.40	0.08	0.963 ns	1.54	0.51	0.78	0.14	0.124 ns
Mongolian										
Campanula punctata	2.44	0.76	1.41	0.27	0.142 ns	0.59	0.23	1.50	0.68	0.163 ns
Delphinium grandiflorum	1.54	0.53	3.31	0.89	0.048 *	1.50	0.72	2.69	0.96	0.279 ns
Kalimeris incisa	29.10	4.73	8.89	1.79	0.000 ***	12.41	3.20	8.65	3.11	0.330 ns
Platycodon grandiflorum	0.08	0.02	0.08	0.02	0.850 ns	0.01	0.01	0.01	0.00	0.977 ns

Scutellaria baicalensis	0.37	0.13	0.85	0.28	0.077 ns	0.06	0.05	0.12	0.04	0.269 ns
Tall canopy										
Shared										
Echinops ritro	3.67	1.19	3.87	1.31	0.904 ns	2.55	1.09	3.90	1.43	0.423 ns
Geranium pratense	4.70	0.93	2.44	0.40	0.012 **	16.85	3.89	8.67	1.72	0.029 *
Sanguisorba officinalis	3.28	0.96	2.10	0.53	0.246 ns	4.65	1.28	3.26	0.92	0.337 ns
Thalictrum aquilegifolium	0.09	0.02	0.05	0.01	0.034 *	0.04	0.01	0.05	0.03	0.674 ns
Veronica longifolia	2.24	0.78	0.77	0.43	0.069 ns	1.84	0.52	0.72	0.34	0.053 ns
Mongolian										
Aconitum carmichaelii	0.22	0.04	0.29	0.04	0.162 ns	0.36	0.09	0.30	0.07	0.531 ns
Angelica sylvestris	0.04	0.02	0.02	0.00	0.148 ns	0.60	0.52	0.03	0.01	0.231 ns
Patrinia scabiosifolia	0.76	0.19	0.39	0.11	0.050 *	0.08	0.04	0.08	0.03	0.978 ns

7.3 Discussion

7.3.1 The effect of depth of sand substrate on overall number of forb seedlings, forb and grass biomass and cover values

The 150mm treatment supported a significantly higher number of forb seedlings in both years (p=0.000). However, the forb biomass appeared similar in 2018 (p=0.670) whereas it became higher in the 150mm treatment in 2019 (p=0.020). This suggests that the 150mm treatment can be beneficial for the establishment of a forb community. However, the process behind this outcome is complex.

Rather than a direct effect of moisture stress on the seedling survivorship and the productivity of biomass, the results indicate the outcome of competitive interactions in different conditions. Regarding evidence that constant water availability is critical for germination (Fay and Schultz, 2009; Hitchmough et al., 2001), it is reasonable to believe that 75mm had supported more seedling emergence in the first growing year as a moisture condition can enhance the germination rate even for the species from dry habitats (Hitchmough, 2017a). However, while the wetter plots had a positive effect on seed germination and initial growth of the young seedlings, the intra and inter specific competition took place sooner. The more fertile condition may favour the vigorous species which have ‘imprecise forage’ (Grime, 2002) rather than the less vigorous species which struggle to take the advantages of the water resource. The productivity would then increase the dominance and impose a severe light stress by the dominant canopies (Keddy, Twolan-Strutt and Shipley, 1997). The condition that initially supported more seedlings lead to a higher competitive mortality (refer to Figure 7.10). On the contrast, the 150mm treatment slowed down the community development especially the dominant species at the early seedling stage. This reduced early competitive elimination and led to a longer coexistence of individuals. This process could explain the results that the 75mm treatment possessed less forb seedlings than the 150mm treatment in April 2018. These significant effects persisted in 2019 (p=0.000). However, the species diversity was not much different (Shannon Wiener Index: 3.08 in the 75mm and 3.04 in the 150mm in 2018; 2.96 in 75mm and 2.92 in 150mm in 2019).

In 2018, where the water availability was higher, grass competition increased. This eliminated more forb seedlings and suppressed the establishment of the forb community, which was associated with more stress tolerant characteristics. The grass coverage was significantly higher in the 75mm treatment than the 150mm treatment in May 2018 (p=0.000). This suggests the more effective competitiveness of the grasses from the seedling stage in the 75mm treatment. The wetter condition eventually supported more grass biomass in summer (p=0.001). Grass appeared to have a lasting competitive effect on the forbs especially inhibiting the small forb seedlings over the entire growing season in the 75mm treatment in 2018. This also enhances the speculation in Chapter 4 that the selected grass species might have been more vigorous in a wetter condition. This showed a different pattern to Hitchmough et al. (2001) that grasses tend to have a better growth in the early stage than the forbs under conditions of extreme moisture stress. This may mainly because that the grasses selected for Hitchmough et al. (2001) are originally found on dry habitats while *Deschampsia cespitosa* tend to be more adaptive in ‘damp soil’ (Ash et al. 1992). Increasing moisture stress can be suggested as an effective way to minimise grass competition where the grasses are associated with the competitor traits. In the 75mm treatment in spring 2019, grasses still had early competitive advantages regarding more coverage (p=0.006). However, as the forb community generally broke through the threshold of grass competition (as discussed in Chapter 4), the wetter condition did not make grasses significantly more productive in summer 2019 (p=0.306 for grass biomass in the 75mm and 150mm treatment). Even the 75mm treatment could only increase grass competitiveness in a short term and presumably was still too stressful for grass biomass production in a longer term. It is reasonable to predict that sowing the same species composition would create a grass dominant community on the existing fertile topsoil on site.

In August 2018, there was no significant difference on forb biomass production between the two treatments (p=0.670). This is an interesting trade-off on the forb biomass production between the two treatment. As the more forb seedlings associated with less resource (predominantly water) availability but where more resources were available led to a greater inter and intra species competition. Neither of the treatment could support more forb biomass production. A different forb biomass scenario was evident in August 2019, with significantly more biomass in the 150mm treatment (p=0.020). This is mostly because of decreased grass competition in the spring 2019 and this changed the dynamic between forb biomass within the 150mm treatment (to be explained in the next section).

Weeds invasion is always a major concern especially in exposed soil when the canopy remains open. The 150mm plots had significantly larger bare ground coverages in both years (16.7% in 75mm and 33.9% in 150mm in May 2018, p=0.000 in 2018; 2.5% in 75mm and 4.1% in 150mm in May 2019, p=0.025). However, the weeds biomass was not significantly different in 2018 (9.37g in 75mm and 7.84g in 150mm, p=0.409), although occasional manually weeding out of big ruderal weeds in early spring in 2018 might be responsible for levelling out the difference between the two treatment. The difference started to be apparent in 2019 (16.84g in 75mm and 12.30g in 150mm, p=0.026). The higher soil moisture treatment was subject to more establishment of weeds biomass. However, as the community was established with a relatively weeds free outset, weeds invasion has low likelihood to be a major concern in practice.

Figure 7.10 Within the same sowing rate, the '75mm' treatment (right) boosted the growth especially of the dominant species from the beginning and this potentially led to both intra and inter species competition taking place sooner (picture taken on the 9th Aug 2017)

7.3.2 The effect of depth of sand substrate on biomass of dominants (*Achillea millefolium* and *Echinops sphaerocephalus*) and subordinates

A high level of species richness can enhance the visual appearance of an urban grassland community. A persistent coexistence of 'stress tolerant' forbs, which are commonly used in designed communities, requires an unproductive soil to reflect their original habitats (Hitchmough et al., 2003). In this study, comparing the treatment effect on the biomass of dominant forbs of *Achillea millefolium* and *Echinops sphaerocephalus* with the overall biomass of subordinates, the 150mm treatment resulted in an increase of dominance with a disappearance of subordinate forbs biomass in the long term.

In 2018, no treatment led to a significantly difference in the biomass of *Achillea millefolium* (p=0.416) and *Echinops sphaeocephalus* (p=0.117). The scenario was changed in 2019 that while the biomass of *Achillea millefolium* still stayed at the similar level in both treatments (p=0.400), but *Echinops sphaeocephalus* developed significantly more biomass in the 150mm treatment than the 75mm (p=0.008). In contrast, the subordinates had a significantly higher biomass production in the 75mm in both years (p=0.000). It is reasonable to assume that the treatment supposed to impose a higher moisture stress did not affect the growth of dominants since those species were less sensitive to the stress and sooner explore the beneath sand fertile resources. However, the growth of subordinate forbs was suppressed by the moisture stress. Despite it supporting a higher initial number of seedlings, the benefit could not compensate the further growth.

The trait characteristics of the dominant forbs facilitated their establishment with the moisture stress. *Achillea millefolium* can be cultivated with a wide range of soil moisture (Ash et al., 1992). This perhaps suggests that the biomass production of *A. millefolium* in summer was less affected by the difference of moisture stress in this system *Echinops sphaerocephalus* took the advantage of its taproot system (as discussed in Chapter 4). The big jump in the biomass in 2019 suggests the time point when the roots started to effectively contact the fertile soil beneath the sands. However, despite no statistical difference, both species had a higher biomass production in the 150mm treatment in 2018 (550.59g in the 150mm and 495.96g in the 75mm for *Achillea millefolium*; 127.91g in the 150mm and 97.13g in 75mm for *Echinops sphaerocephalus*). This suggests evidence of suppression from the grasses and early emerged subordinate forbs in the 75mm treatment.

The 150mm treatment positively affected the biomass of *Echinops sphaerocephalus* and produced significantly more biomass than the 75mm treatment (p=0.008) in 2019. The predominant reason was the 'tall grass' competition which significantly affected *Echinops sphaerocephalus* biomass in 2019 (as explained in Chapter 6) was greater in the 75mm treatment from spring (more grass coverage, p=0.006). This was explained by the interactive results of 'depths of sand x grass height' in the 150mm treatment significantly increasing *Echinops sphaerocephalus* biomass in the 'tall grass' treatment (p=0.002) but not the 'short grass' treatment (p=0.253). No statistical difference in the treatment of 'depths of sand x heights of grass' for *Achillea millefolium* biomass again suggested that *Achillea millefolium* effectively escaped the grass competition of both types of grasses. Besides, the competitive subordinate species that benefitted in the 75mm treatment might also have decreased the biomass production of *Echinops sphaerocephalus*. Lastly, the 150mm accommodated more *E. sphaerocephalus* seedlings in 2019 (3.4 in the 75mm and 4.2 in the 150mm treatment). This might have affected the biomass production as the individuals were larger, however, this was unlikely to significantly change the biomass pattern.

The higher availability of moisture resource in 75mm treatment benefitted the growth of subordinates and potentially made them more competitive in at least some subordinates. It is also worth mentioning that while the grasses were more competitive in the 75mm treatment, these 'competitive subordinates' also showed tolerance of grass competition where a higher level resource is in place. The competitiveness of subordinates, that especially had faster growth in the early stage, might to some degrees negatively affect the production of dominants in the 75mm treatment. As Grime (2002) suggests that the persistence of dominance depends not only upon the generation of stresses to subordinates but also upon the capacity to avoid or resist the subordinates effects. Subordinate biomass in the 75mm treatment has higher potential to be enhanced by management approach such as a mid-spring cutback or dominant species removal. Introducing severe moisture stress may however cause a significant decline in subordinates but less affect the dominants. The loss of subordinate biomass may lead to drift to monoculture. Gaining competitiveness for subordinates is essential to coexist with dominants.

Table 7.4 Categories of effects (positive/ negative or no effect) of depth of sand substrate on dominant, subordinate and grass biomass

Does increased moisture stress affect the abundance biomass?	*Achillea millefolium*	*Echinops sphaerocephalus*	Subordinate forbs	Grasses
2018	No effects	No effects	Negative effects	Negative effects
2019	No effects	Positive effects	Negative effects	Negative effects

7.3.3 The effect of depth of sand substrate on the number of seedlings and biomass of subordinate species

In both years, none of the subordinate forbs had a significantly greater number of seedlings in the 75mm treatment; and none of the them produced significantly more biomass in the 150mm treatment. Again, this is a two-way response of resource availability and competition stress in the performance of subordinates. In spring the species that had a significantly lower number of seedlings in the 75mm treatment suggesting greater mortality from the dominants' competitive elimination, and especially grass competition. In summer, the higher biomass produced in the 75mm treatment mostly suggested species were sufficient competitive to coexist with grasses under a relatively higher fertile condition. Both of these patterns of seedling number and biomass remained similar in both 2018 and 2019. Species with ability to actively compete for light again tend to be successful in both seedling number and biomass abundance.

In terms of subordinate seedling number, the risk of being eliminated reduced with the increase of forb canopy height in both years. This was especially so in the designed lower canopy species which were universally shade intolerant. Seven in 2018 and 6 in 2019 out of 10 low canopy species had significantly lower seedling number in the 75mm treatment, while the number decreased to 3 in 2018 and 2 in 2019 out of 6 for the medium canopy species (*Scutellaria baicalensis* and *Platycodon gradiflorum* data in 2018 and 2019 were not tested due to inconsistent availability of valid data) and 2 out of 6 in 2018 and 1 out of 8 in 2019 (*Aconitum carmichaelii* and *Angelica sylvestris* data were not validated in 2018 due to the same errors). Besides, the tall canopy species used in this study had relatively high comparative rates of biomass production (regarding Table A in Appendices). Advantages of biomass accumulation in the early stage appeared to be essential to persist in community competition (Tremmel and Bazzaz, 1993; Torner *et al.*, 2000).

On the contrast, the species that were not sensitive to the difference in the treatment appeared to have stronger abilities for light competition. Those abilities included the architectural form of clambering stems (e.g. *Galium verum*), tall leafy stems (e.g. *Kalimeris incisa* and *Veronica longifolia*) and elongating leaf petioles (e.g. *Geranium pratense*). Moreover, early emerging and rapid recovering species such as *Potentilla rupestris* also appeared persistent. Also, *Origanum vulgare*, *Polemonium caeruleum* and *Sanguisorba officinalis* that had both advantages of leafy stems and early biomass production were also persistent in both treatments in both years.

The subordinate biomass was generally less sensitive than the number of seedlings, and the pattern of subordinate biomass appeared to relate less to the species canopy height. However, similarly, the advantages in the architectural form and early emergence were still beneficial to produce more biomass in the 75mm treatment. The competitive subordinates in both years were *Galium verum*, *Potentilla rupestris*, *Dracocephalum ruychiana* (short leafy stems), *Veronica teucrium* (with some leafy stems), *Origanum vulgare* and *Geranium pratense*.

The low canopy species were likely to be more tolerant to moisture stress while more tall canopy species were more adaptive to the wetter conditions (according to the Ellenberg's indicator values of moisture in Table A in Appendices). This might affect the pattern that the low canopy species had more seedling numbers in the 150mm treatment. However, the 75mm plots did not benefit from more seedling for the tall subordinates. Also, the subordinate biomass was independent of the pattern of moisture adaptiveness. This again suggests that the moisture adaptiveness was not directly involved to shape the community performance but the light competition still played the major role (Köppler and Hitchmough, 2015).

Chapter 8 Final discussion and conclusion

8.1 Final discussion

8.1.1 Do grasses inevitably dominate sown meadows leading to a decline in forb survival and biomass?

Ecological studies suggest that grasses are the superior competitors and forb dominance is mostly transient in fertile experimental grasslands. On the other hand, it is clear that there are many examples of high forb content in many long term semi-natural and designed meadow landscapes. Where grasses are left out of designed communities, weeds tend to become problematic leading to a decline in forb performance and increasing the management input from the beginning (Hitchmough, Paraskevopoulou and Dunnett, 2008; Dickson and Busby, 2009; Hitchmough, 2009). In the study described in this thesis the first question was to what extent is it practicable that a sowing mix can contain some grasses but does not lead to competitive elimination of the forbs in at least the short term.

In designed planting projects, grass competition is hostile to the establishment and persistence of new forbs in productive soil (Hitchmough and Fleur, 2006; Hitchmough, Paraskevopoulou and Dunnett, 2008; Hitchmough, 2009). Similarly, disturbance such as cutting back at specific times or graminicide treatment did improve forb performance in terms of forb seedling density, species richness and biomass but this did not decrease the competitiveness of grass which potentially leads to dominance (Hitchmough, Paraskevopoulou and Dunnett, 2008). However, by designing the outset, grass dominance can be voided for a much longer period. Design approaches such as topsoil removal and sand mulch layer application can effectively reduce weedy grass emergence from the soil seed bank or seed dispersal from outside, and the soil treatment used at the establishment stage is more important than the later management treatment (Hitchmough and Fleur, 2006). First year establishment involving a high starting high biomass ratio of desired forb: weedy grass can lead to at least 5 years forb dominance (Hitchmough and Wagner, 2013). This can be achieved by sowing in an optimal season (Hitchmough and Fleur, 2006) and in some cases weeding or irrigation to improve establishment (Hitchmough, 2017a). Moreover, the design process on species composition including 1) density, 2) layering and 3) species trait characteristics can facilitate communities that do not naturally co-occur (Hitchmough, 2017b; Hitchmough, Wagner and Ahmad, 2017). High sowing density can increase the initial ratio of designed forbs: weedy grass (indicated by cover values) (Hitchmough and Fleur, 2006); multi-layered communities can extend the vegetative covering period and occupy more niches reducing weed colonisation (Hitchmough, Wagner and Ahmad, 2017); using shade tolerant forb species or productive tall leaf forb species which are broadly equivalent to the existing grasses can increase forb community persistence (Hitchmough, Paraskevopoulou and Dunnett, 2008; Hitchmough, 2009) even with non-native species (Hitchmough, 2017a; Hoyle *et al.*, 2018).

The practicability of achieving forb rich communities in the longer term has been shown in a series of urban meadow communities that have remained dominated by forbs for almost ten years for example Queen Elizabeth Olympic Park, London, Native Wildflower Meadows and Fidelity International (Tonbridge) native meadows (as shown in Figure 8.1 and 8.2) (Hitchmough, 2017a). Both of these commenced with much reduced grass components in the seed mix and a surface layer of sand sowing mulch to restrict grass emergence from a soil seed bank. Despite that the capital costs may be high and the first year management may be intensive, designed meadows are more financially sustainable than conventional urban plantings whose maintenance needs increase with passage of time (Hitchmough and Dunnett, 2004). Also, the additional cost is small compared to relatively inexpensive hard landscapes such as pavement and street furniture which are often incorporated in a landscape project.

Figure 8.1 Forb dominance is achieved through design can persists in the longer term with low management input. Top - Queen Elizabeth Olympic Park UK Native Wildflower Meadows. Pictures were taken in July 2012 (top left) and in June 2017 (top right). Bottom - Fidelity International native meadow. Pictures were taken in late July 2013 (bottom left) and in mid-July 2018 (bottom right) (Hitchmough, no date)

Figure 8.2 The same area in Queen Elizabeth Olympic Park UK Native Wildflower Meadows. Zooming into smaller scales, despite grass invasion and colonisation were evident as a part of the ecological process, designed meadows persist high landscape values in the longer term. Pictures were taken in February 2014 (top left), March 2020 (top right) and June 2014 (bottom) (Hitchmough, no date)

Bjørn, Weiner and Ørgaard (2016) argued that a self-managed designed forb dominant community cannot persist in the long term and is gradually dominated by grasses and colonised by woody species. This is theoretically true however urban greenspace with no management is relatively uncommonly. In urban landscapes, greenspace management is generally culturally and politically required to deliver the essential cues of intention and care (Hoyle, Hitchmough and Jorgensen, 2017a) where biomass and dead material are removed post flowering or before turned messy (Hoyle *et al.*, 2017). This process is often considered as a low management approach in urban plantings but permits forb dominance for a long term. A sown community of tall leafy stem prairie species sown in 2004 is still largely weed free in the Sheffield Botanical Gardens because of the intense within canopy shading.

In the author's study, forbs rather than grasses became the competitive dominants, suggesting that life form characterisation is less important than the characteristics of given species. Grass competition negatively affected forbs from the first growing year and significantly influenced the establishment and composition of the forb community in the longer term. However, the grass biomass declined in the third year (2019) and the community moved from 'grasses and *Achillea millefolium* dominance' to '*Achillea millefolium* and *Echinops sphaerocephalus* dominance'. This differs from the conclusion that grasses are always superior competitors that constantly reduce forb richness and biomass in a grassland system (Pywell *et al.*, 2003; Del-Val and Crawley, 2005; Dickson and Busby, 2009). Inevitably the selection of grass species in the study determined the intensity of grass competition experienced by the forbs. As the study was focused on competition within the sown mix rather than competition between the sown mix and species invading from the outside, *Deschampsia cespitosa* was selected. This grass was chosen for a number of reasons; firstly, the structure and visual appearance of this species is more favourable in landscape design contexts than many C3 grasses in both Europe and Mongolian. This species has upright foliage and tussock form and retains this structure for a longer period post flowering. Its tussock structure and reduce initial growth rates potentially offered the potential not to eliminate all sown forbs in the short term and its tussock form might provide some resilience to invading grasses. Secondly this species is commercially available in both short and tall forms thus representing a grass competition gradient without having to employ two different species present which would involve more experimental variables. In terms of competitive the tall form of *Deschampsia cespitosa* was expected to be as competitive to forbs as the dominant weedy grasses *Holcus lanatus* and *Lolium perenn*e, but not as competitive as *Arrhenatherum elatius*. These grass species were also the main invaders in the studies in the UK context such as Del-Val and Crawley (2005) and Hitchmough, Paraskevopoulou and Dunnett (2008). Species trait characteristics suggest, *Arrhenatherum elatius* develop deeper roots and are naturally found with low forb species richness than *Deschampsia cespitosa*; *Holcus lanatus* have higher relative growth 1.56 g/g/w (1.45 g/g/w for *Deschampsia cespitosa*) and adapt to wider range of habitats; and *Lolium perenne* develop denser and more lateral tiller which is more likely to shade out the neighbours (Grime, Hodgson and Hunt, 1988; TRY Plant Trait Database, 2020). *Deschampsia cespitosa* grow more slowly in the seedling stage than the other weedy grasses, although to some degree this is compensated for by the formation of a robust, permanent structural tussock in *Deschampsia cespitosa* (Hitchmough, unpublished). The shorter form of *Deschampsia* is more akin competition wise to vigorous forms of *Festuca rubra.* This study predicts different outcomes if any of those grass species were used in the study.

Some of the forbs used in this study have greater capacity for dominance than those used in the studies of Del-Val and Crawley (2005) and Hitchmough, Paraskevopoulou and Dunnett (2008). The forb biomass was predominantly represented by the biomass of *Achillea millefolium* in 2018. Grass effects on *A. millefolium* determined overall forb biomass under grass competition. *A. millefolium* showed a similar level of competitiveness to *Deschampsia cespitosa* competition and this suggests a roughly symmetric pattern between forb and grass biomass competition (Pearson's correlation efficient = -0.709, p=0.000). By 2019, *A. millefolium* seemed to have escaped grass competition as the tall leafy stems effectively penetrated though the grass canopy. *Achillea millefolium* adapts to a wider range of habitats than *Deschampsia cespitosa*, with the latter more sensitive to moisture stress (to be discussed in later part) (Grime, Hodgson and Hunt, 1988; Hurteau, 2003; St. John, Ogle and Darris, 2011). *Achillea millefolium* can colonise bare ground more effectively than many forbs and even some grasses (Bjørn *et al.*, 2019). This indicates competitiveness and ability to dominate from emergence stage in sowing communities. Dwyer (1958) found rhizomatous forbs significantly decline grass biomass in a community by vegetative expansion. In this study, the rhizomatous growth form of *A. millefolium* was likely to be optimised in sand substrates where are more penetrable than clay based soils. Apart from *A. millefolium, Echinops sphaerocephalus* emerged as one of the earliest species in the year and were observed as one of the most rapid growing forb, producing a much taller foliage above grass canopy. Also, *E. sphaerocephalus* became more productive when the taproots contacted to the fertile soil beneath the sand mulch in 2019. This agrees with Dwyer (1958) that forbs with taproot system are also potentially competitive to grasses. The experimental conditions allowed both *A. millefolium* and *E. sphaerocephalus* to maximise their productivity.

Using productive forbs that are broadly equivalent to grass competitiveness can help retain a community forb dominant (Hitchmough and Fleur, 2006; Hitchmough, 2009). Without these two species in this study, grasses would be more dominant and more persistent in the longer term. However, some other forb species such as *Origanum vulgare, Geranium pratense* and *Sanguisorba officinalis* are also able to coexist with vigorous grasses and persist in the longer term. This is because they can, in various ways project their leaf canopies above the grass foliage, and naturally found in relatively productive grasslands. This again suggests competition for light is critical if shade intolerant forbs are to compete with grass biomass (Pywell *et al.*, 2003; Del-Val and Crawley, 2005; Hitchmough, Paraskevopoulou and Dunnett, 2008).

Moisture stress created by the sand mulch treatment effectively benefitted forb establishment in the system. Moisture stress is likely to have restricted growth of grass seedling of this moisture demanding species but less affect the growth of relatively unproductive forbs. This would help increase the starting biomass ratio of designed forb: grass and this initial establishment can be a good indicator of successful forb performance in long term (Hitchmough and Fleur, 2006; Hitchmough and Wagner, 2013). This also agrees with the review of ecological restoration success in Pywell *et al.*, (2003) that forb richness is more likely to persist in unproductive sites. More importantly, the sand mulch treatment minimised the potential impacts of weedy grass competition from the underlying soil by restricting the capacity of the grass seed bank and growth which were identified as key advantageous traits of dominant grasses by Pywell *et al.*, (2003). Practically, the model of establishing relatively unproductive forbs in sand substrate (at least a depth of 75mm) on top of productive soil has been shown to support forb dominant communities beyond the initial establishment period (Hitchmough and Wagner, 2013). Moreover, sand mulch layer also reduces herbivore damage from molluscs (Hitchmough and Fleur, 2006) which more negatively affects forbs than grasses (Edwards and Crawley, 1999a; Del-Val and Crawley, 2005), although not all of the forb species in this study are palatable. This emphasises the importance of preconditions for the ecological process that determines the community performance (Walker *et al.*, 2004) and design of outset conditions for landscape planting projects in the long term (Hitchmough, 2017a). The severe drought and heat in summer 2018 also limited the grass competitiveness, especially in this study with *Deschampsia cespitosa* naturally being associated with moisture conditions (Grime, Hodgson and Hunt, 1988; St. John, et al., 2011). The summer in 2018 was the UK's warmest since 2006 and the driest since 2003 (about 50 dry days on average across England) (Met Office, 2018), with 23 continuous dry days across from June to July in Sheffield (Sheffield Weather Page, 2020). The level of rainfall can affect the biomass ratio of forb: grass within a community until the following year (Silvertown *et al.*, 2006). Spring 2019 was again dry and hot again reducing grass competition, despite the weather was not being as extreme as summer 2018. To retain more seedlings, the experimental field was manually irrigated every ten days in absence of rain in summer 2017 and 2018.

In this research, provided two contrasting levels of moisture stress and nutrient availability from the sand substrate by using different depths to reduce the competition of grass species which are more sensitive to moisture stress and nutrients than selected forbs. Biomass of the successful forb species that developed leafy foliage above grass biomass before grasses fused the ground were also benefitted. The sand mulch layer effectively reduced ruderal weeds establishment and competition from external seed rain and emergence from soil seed banks in the underlying productive soil. The weed biomass as a percentage of sown species biomass was extremely low at 0.71% in August 2018 and 0.78% in August 2019 although this is also due to the high sowing density. In the later stage, grass closed the ground layer to maintain weed resistance. This process to preventing weed invasion was effective considering that the site had intensive cultivation history and an abundant weedy seed bank; and was surrounded by uncut invasive grasses. Despite hand weeding of tall ruderal weeds until May 2018, management input is relatively low. During the experimental period, the community was managed by summer (August) and early spring (February) cutback and biomass removal. This also facilitated shorter canopy forbs to photosynthesise increasing the likelihood of survival and growth.

The Forb dominant community is likely to be retained in long term as the successful forb species are established and likely to compete with *Deschampsia cespitosa* in terms of light competition whilst, forb seedling numbers will continue to decrease in the next few years until an equilibrium is reached. Hitchmough and Fleur (2006) observed that the seedling survival tends to become more stabilised from the fourth year. This might may also apply to the author's study. It was not possible to retain the experimental plots in the longer term, however this study anticipated that grass biomass would mostly retain a similar level of biomass production in the future. *D. cespitosa* were observed to have the characteristics of early emergence and rapid recover after biomass removal, key requirement to persist in the community (Bullock *et al.*, 2001; Pywell *et al.*, 2003). This is also an important functional role to prevent weeds invasion and an aesthetic role of vegetative cover in unproductive conditions. The author's study anticipates that there would be ongoing colonisation of the communities by other grass species, although it seems likely that these would struggle to compete with *Deschampsia cespitosa* and the dominant forb community. Besides, grass biomass threshold to reduce forb biomass appeared lower in 2019 than 2018 (to be explained below) perhaps indicating an increasing grass competitiveness with time. St. John et al., (2011) indicates that *D. cespitosa* can take a few years to establish; also, as a greater proportion of the grass roots establish in the moist fertile soil beneath the sand mulches, *D. cespitosa* may reassert itself. Both suggests that grass have potential ability to bounce back in the future. Both the grass and dominant forbs showed adequate competitiveness in the system. This study predicts the coexistence will last for a much longer term but the biomass equilibrium between forb and grass may still largely depend on both climate and management approaches.

Grass competition significantly reduces seedling number and biomass of either less fitted forb species or those with disadvantageous traits of seedling growth rate and shoot architecture. From the third year (2019), grass competition appeared less critical to forb biomass. Although the community became ‘forb dominated’ in 2019, plots which were associated with high number of grass seedlings had substantial grass coverage in spring and this continued to eliminate slow growing, low, and late emerging forb seedlings by light competition. The low canopy forbs in this study are mostly shade intolerant (refer to Ellenberg’s indicator value of light, Table A in Appendices). Using shade tolerant forbs in the understorey layer would enhance forb richness in the subordinate group according to Hitchmough, Wagner and Ahmad (2017) and this study suggests this approach in the future practice where the aim is not to mirror a naturally occurring community.

Deschampsia cespitosa ‘Barcampsia’ (the ‘tall grass’ in this study) were re-sown in December 2017 and the seedlings were transplanted in May 2018 to counteract low emergence of the grass in this treatment in 2017 (as explained in the Methodology). This to some degree simulated the universal problem of weedy grass invasion to a forb-only sowing or turf community (Pictorial Meadows, 2020) from the second growing season; particularly in the fertile topsoil or with abundant weedy seed bank. In this research, ‘Barcampsia’ establishment was low in 2018 mostly because the competitive forbs were more established and imposed light stress to grass seedlings. However, the planted ‘Barcampsia’ (in summer 2018) produced more biomass in 2019; they were advantaged by their quick recovery post cutback in August 2018 and early emergence in spring 2019. This process did not affect the overall forb dominance which was predominantly imposed by the biomass of *Achillea millefolium* and *Echinops sphaerocephalus*. The author’s study suggests that a forb dominant community that resistant to weedy grass invasion can be achieved by using competitive forbs that can maximise forb productivity in the certain condition, but that these forbs will exert a negative effect on subordinate forbs in the same way that grass does (Hitchmough, De La Fleur and Findlay, 2004; Hitchmough and Fleur, 2006; Dickson and Busby, 2009).

To speculate on the long term, for example the next 20 years, community succession will largely depend on self-seeding. This study predicts that most of the forb species would be outcompeted by interspecific competition of grasses or dominant forbs before the time point. Depending on seed dispersal only is unlikely to achieve diverse forb communities in the longer term, especially in fertile conditions (Davies, Dunnett and Kendle, 1999; Hitchmough, 2000; Sluis, 2002; Dickson and Busby, 2009) while low fertile soils such as sands reduce interspecific competition but may restricts desired species colonisation by self-seeding (Hitchmough and Fleur, 2006). The ability of forb rich community to persist for decades only with limited management is unlikely to be politically required in most designed urban landscapes. Communities maintained by early August cutting and hay removal have higher capacity to persist. However, an autumn or early winter cutback to extend the landscape values may be required in the long term in practice for the public. This however may encourage grass seedling establishment at the expense of some forb species (Pywell *et al.*, 2003; Dickson and Busby, 2009; Bjørn *et al.*, 2019). This study argues the importance of creative management strategies to retain forb richness in designed meadows (Hitchmough, 2017a), despite this not being the main focus in this study.

8.1.2 Is there a critical (grass) biomass threshold for decline in forb biomass?

To our knowledge, there are no precedent studies on critical values of grass biomass for retention of forb biomass. The biomass guidance across productivity gradients for either ecological restoration or planting design remain unanswered. In this study, the original objective was to test the grass biomass thresholds on forb retention in 2018 and 2019. However, the forbs in this study, mainly *Achillea millefolium* and *Echinops sphaerocephalus,* were too vigorous in the system and no grass treatment led to a reduction in their biomass, and grass biomass actually declined. Thus, the original hypothesis was not able to be tested. The grass biomass thresholds were tested separately in 2018 and 2019 to identify the equilibrium points at which grasses begin to dominate.

All 96 plots in this study were ranked from the lowest to the highest grass biomass volume in a row. Linear trendlines for ranked grass biomass and associated forb biomass were then plotted to calculate the cross over point between both. In 2018, the crossover appeared at 471.89g of grass biomass (per 800mm x 800mm quadrat) (refer to Figure 8.3). This suggests that grass biomass beyond this point began to reduce forb biomass. In 2019 there was no crossover between the two trendlines, they had to be extended to forecast the threshold according to the tendency. The grass biomass threshold was predicted at 441.84g (refer to Figure 8.4). This indicates a lower grass biomass threshold to reduce forb biomass in the community. Forb biomass might be more sensitive to grass competition in the longer term and this agrees with (Sluis, 2002; Pywell *et al.*, 2003; Dickson and Busby, 2009). The advantages of grass morphological characteristics for example quick recovery after cutback and growth of dense and upright foliage allow grasses more effective to dominate physical space and light resources. This can enhance the impact of grass growth and accumulate competitive advantages over most forbs in the community in the longer term. Besides, the virtue of being less palatable to molluscs can also be increasingly essential to maintain competitiveness in the developing community (Edwards and Crawley, 1999a; Del-Val and Crawley, 2005).

However, this study shows that large biomasses whether of grasses or forbs do reduce the density and diversity of subordinate forbs.

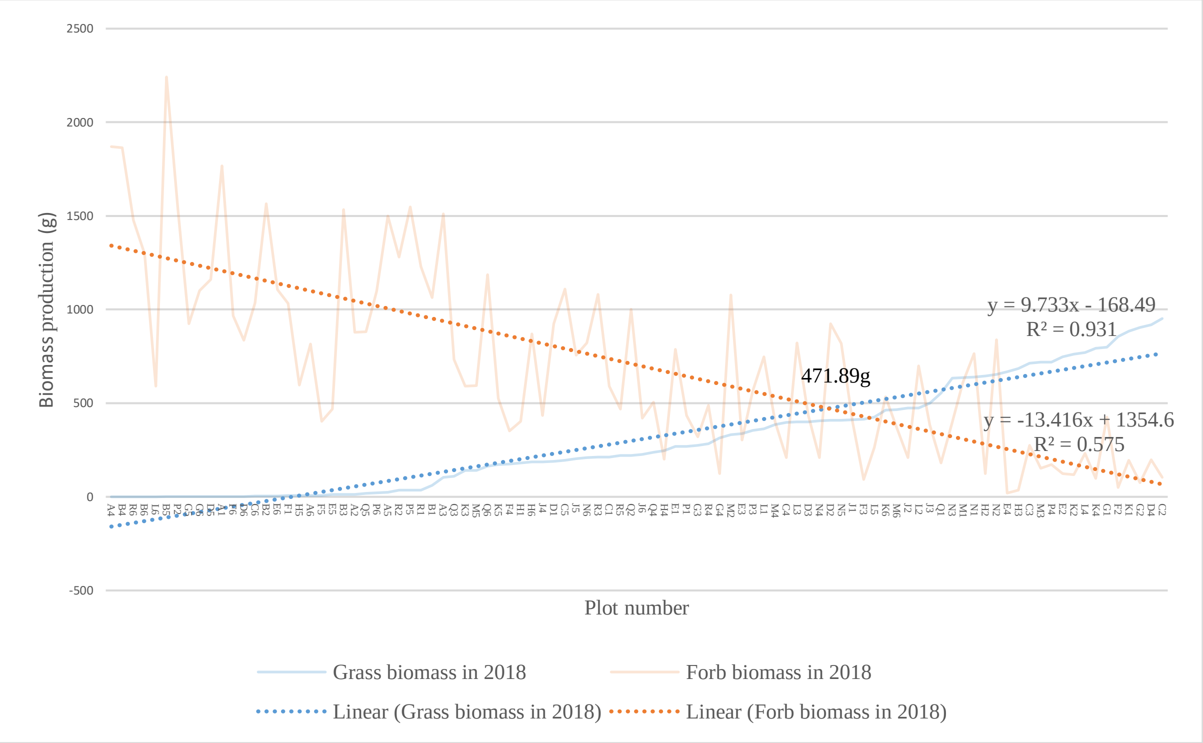


Figure 8.3 Grass biomass threshold on declines of forb biomass in 2018

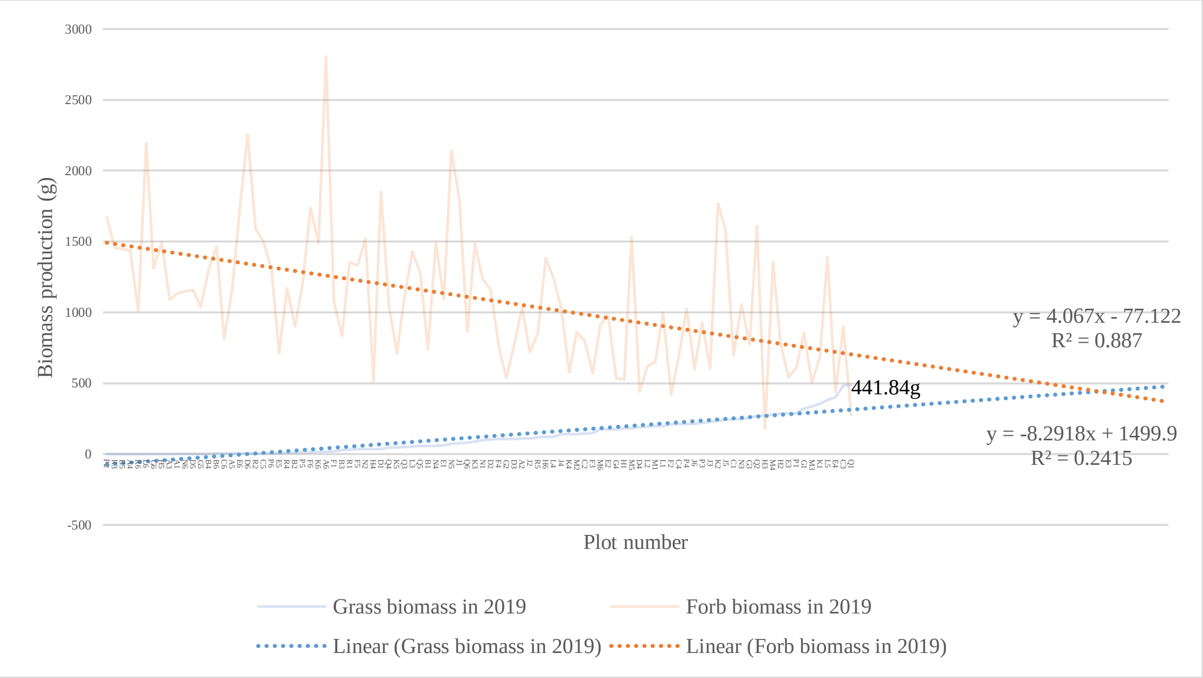


Figure 8.4 Grass biomass threshold on declines of forb biomass in 2019

8.1.3 To what extent, can the design approaches enhance forb performance against grass competition?

Although the treatment effects gradually reduced over time, due to competitive elimination from grasses and more dominant forbs, and this study anticipated that the forb seedling number will reach a similar level in few years. All designed treatment increased the number of forb seedlings in at least three years by increasing forb sowing density, decreasing grass sowing density and increasing moisture stress (particularly, as in this study when the grass is more sensitive to moisture stress than many of the forbs). This agrees with previous studies of for example Hitchmough, Paraskevopoulou and Dunnett (2008) and Dickson and Busby (2009). However, these treatments were much less effective on forb biomass from the second year (2018). Statistical model showed that the only effective treatment on the forb biomass for the first two years (2018 and 2019) results were 'the sowing ratio of forb: grass' (p=0.000). This was predominantly because that the treatment involved extreme starting points between forb and grass (nine times more forbs to grass, and vice versa).

An increased forb sowing density did not always increase forb biomass even in the short term (from 2018) but this depended on the context of grass competition. The high forb sowing density only increased forb biomass in the interactive treatment of 'high sowing density x tall grass (*Deschampsia cespitosa* 'Barcampsia')' and 'high sowing density x 75mm depth of sand mulch (the shallower sand layer)' in 2018. This may because the tall grass had very low emergence comparing with the forb group in the initial sowing. Increasing sowing density for both led to increase more forb emergence and biomass. Whereas the emergence of grasses was higher for the 'short grasses', increasing the sowing density merely increased the symmetric competition between grass and forb, and resulted in the similar biomass pattern between forb and grass. Moreover, compared with the '75mm sand substrate' treatment, the '150mm' treatment discouraged grass competition while the forbs which were less sensitive to moisture stress could dominate the community and reach the maximum biomass productivity at a relatively low sowing density. This agrees with Stevenson, Bullock and Ward (1995) who recommended a relatively lower forb sowing density to be used in unproductive conditions and where competitive perennial grasses are both less productive and likely to be present at lower density. The author's study suggests that the approach of increasing forb sowing density is effective in the situation where grass competition is intermediate but not high or low.

Sowing density had no significant effect on forb biomass (but did increase seedlings in the short term) in any interactive treatment in 2019, in practice having a higher number of seedlings and biomass at the initial stage can close the ground to prevent weeds invasion and establish visual effects sooner. This is essential to increase the likelihood of forb community persistence in the longer term (Stevenson, Bullock and Ward, 1995; Hitchmough and Fleur, 2006). However, this approach primarily increases the establishment of vigorous species and bring forward the effect of dominance especially under productive conditions (Keddy, Twolan-Strutt and Shipley, 1997). This leads to a greater self-thinning which result in a reduction of less vigorous species (e.g. slow growing species). This dilemma has always been a core issue among many landscape practices. Future research should aim to identify critical threshold densities for the persistence of a range communities and species at different productivity levels (Hitchmough, 2017b).

In terms of designed seedling ratio of forb: grass, the threshold to cease forb biomass increase was identified at the ratio of 1: 1. Grass competition was suppressed at this point, and adding more forb seedlings could not increase forb biomass but only increase inter and intra species competition beyond this point from 2018. In practice, adding more forb seeds to enhance forb biomass would become cost ineffective beyond a certain threshold. Again, this threshold largely depends on the level of grass competition. With limited grass competition, the forb biomass may represent most of the productivity irrespective to the initial sowing rate. However, this pattern would be reversed where conditions better fit the growth of grass species. In this study, the ‘75mm sand substrate’ enhanced grass competitiveness by facilitating root access to the moist soil beneath the sand and the advantage of having higher seedling ratio of forb: grass was evident that the ‘50% forb ratio’ produced 700.58g while ‘90% forb ratio’ had 951.23g forb biomass (despite no statistical significance). This study also predicted that grass biomass could dominate the community irrespective to the designed ratio if invasive grass species were used or the soil fertility was extremely high. Thus, similar to the suggestion for ‘sowing density’, this study suggests that a high ratio of forb: grass can be effective in a longer term in the scenario where grass competition is substantial but not severe.

Due to the significant difference in grass seedling emergence between the ‘short’ and ‘tall’ grass treatments from the beginning (as explained in Chapter 6), higher grass competition effects where grass emergence was greater eliminating forb seedlings and reducing forb biomass establishment were evident in 2018 as has been reported by Del-Val and Crawley (2005). However, the ‘short’ grass biomass threshold for declines in forb biomass appeared at 496.08g (as shown in Figure 8.5) while the ‘tall’ grass biomass threshold appeared much lower at 392.10g (as shown in Figure 8.7). This indicated a higher competitiveness of the ‘tall’ grass biomass. This is presumably due to the faster growth in spring and the taller foliage in summer being more effective at suppressing the forbs.

Despite a uniform grass coverage achieved between the ‘short’ and ‘tall’ grasses by spring 2019 through manipulation of new sowings and mainly seedling transplantation, overall forb biomass showed no significant differences between the two treatment. This suggested an interesting trade-off between the forb biomass under lower light competition from the beginning and the forb biomass under higher light competition but from the ‘mid-term’ that allow the forbs to be more established. The better establishment of forbs in early years appears important to enhance competitiveness in the longer term (Hitchmough and Fleur, 2006; Hitchmough, 2017a). The grass biomass threshold below which forb biomass was greater was higher in the ‘tall grass’ (472.83g as shown in Figure 8.8) than the ‘short grass’ (407.74g as shown in Figure 8.6). However, this study anticipates that in the future, forb biomass will be lower in the ‘tall grass’ than the ‘short grass’ treatment from the fourth year, as the ‘tall grass’ showed advantage in light competition in both spring and summer.

As discussed in the previous section, the deeper depth of sand mulch layer reduced grass competition and supported higher number of forb seedlings. However, the lower grass treatment did not increase forb biomass in 2018. Also, despite the forbs in the community appeared more moisture stress tolerant than the grasses, forb growth was also reduced by the greater moisture stress in the ‘150mm’ sand treatment. The grass biomass threshold (below which forb biomass was greater) was 437.43g in the ‘150mm’ (as shown in Figure 8.11) and 502.27g in the ‘75mm’ (as shown in Figure 8.9) treatment suggesting reduced forb competitiveness in the ‘150mm’ treatment. The pattern was reversed in 2019 when the grass biomass threshold in ‘150mm’ treatment was 641.14g (as shown in Figure 8.12) comparing with 348.17g in the ‘75mm’ treatment (as shown in Figure 8.10). This was mainly because that the dominant forbs effectively overcome the moisture stress while the competitiveness of grasses was reduced in the ‘75mm’ treatment.

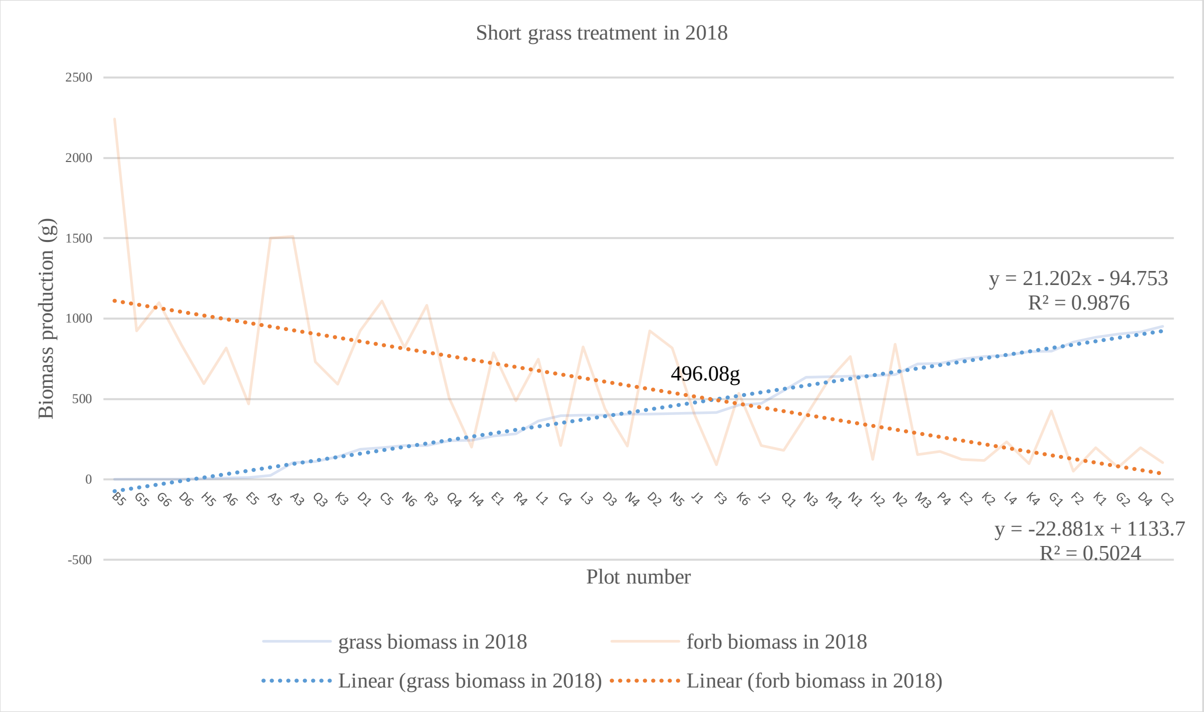


Figure 8.5 Grass biomass threshold within Short grass treatment on declines of forb biomass in 2018

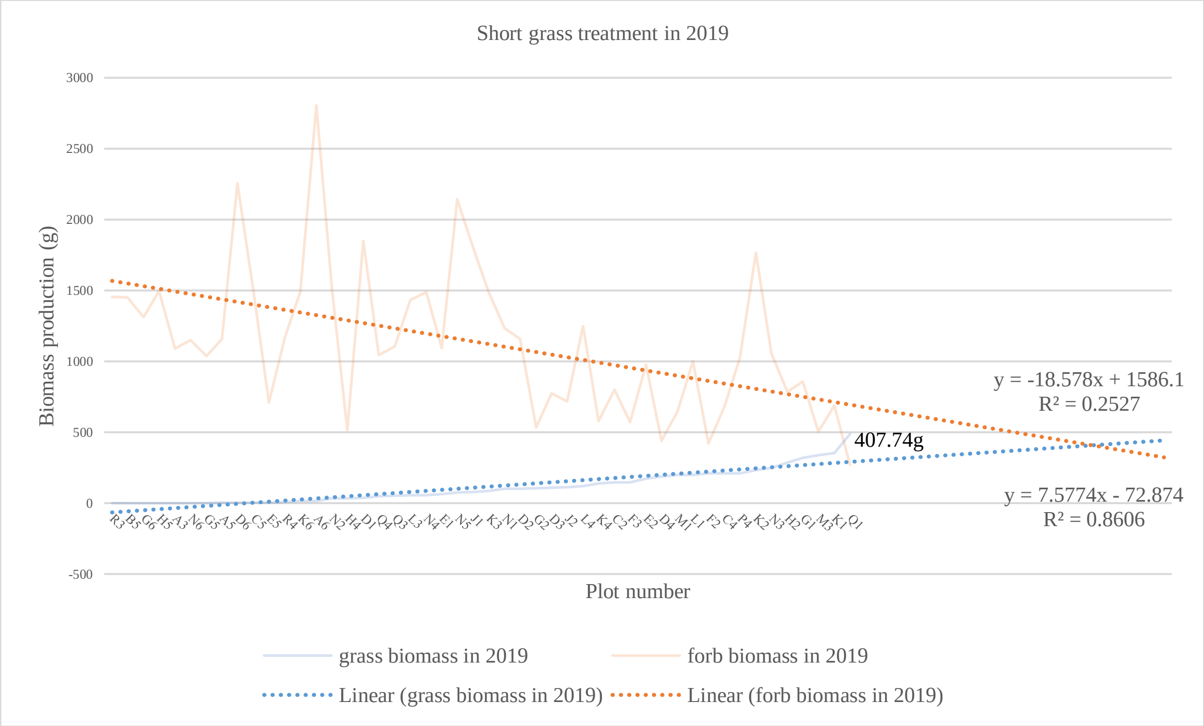


Figure 8.6 Grass biomass threshold within Short grass treatment on declines of forb biomass in 2019

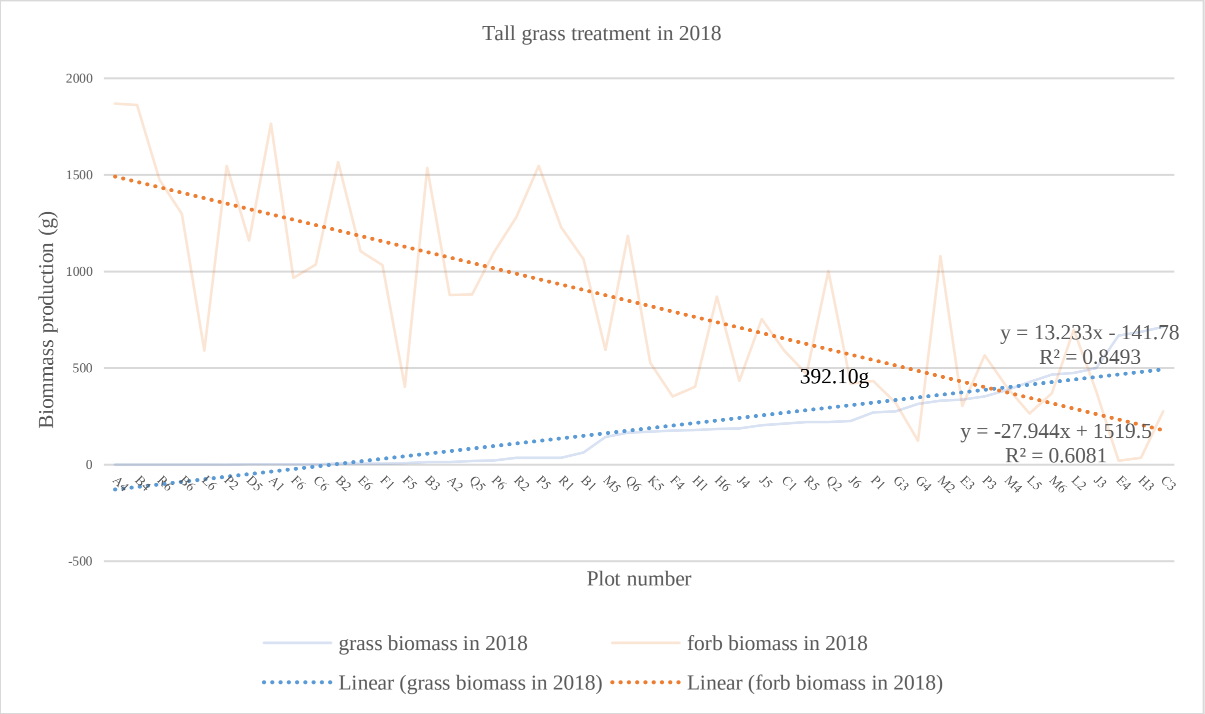


Figure 8.7 Grass biomass threshold within Tall grass treatment on declines of forb biomass in 2018

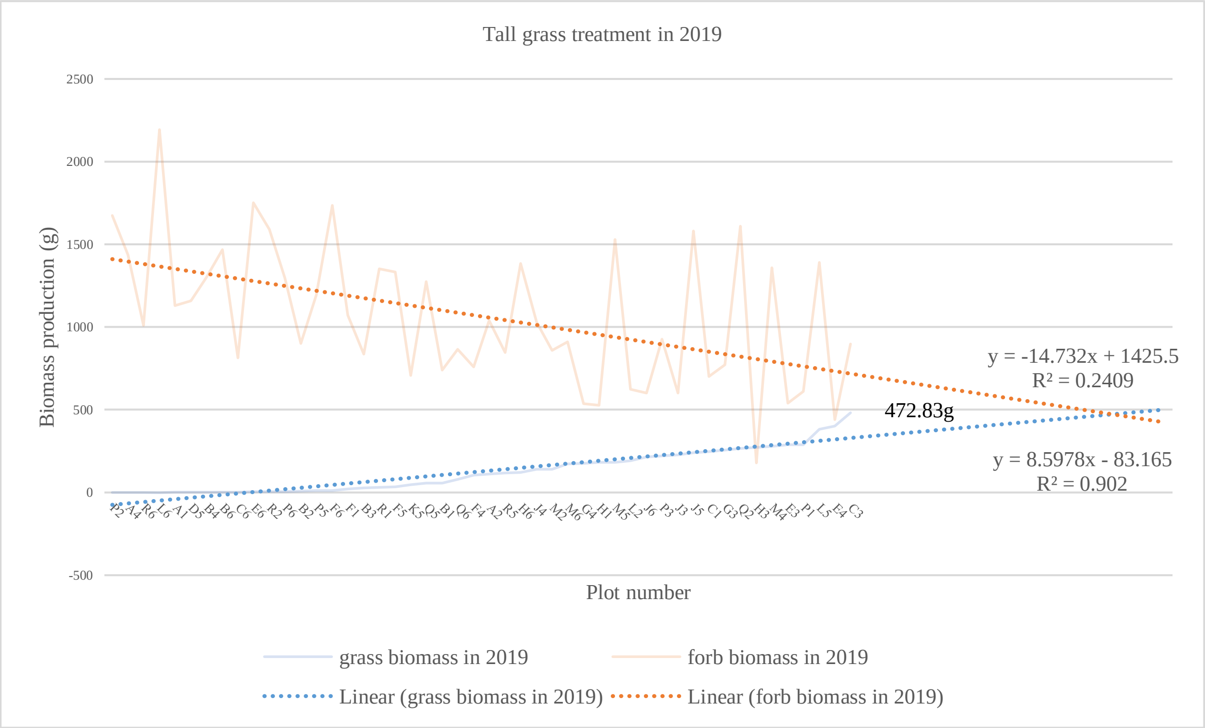


Figure 8.8 Grass biomass threshold within Tall grass treatment on declines of forb biomass in 2019

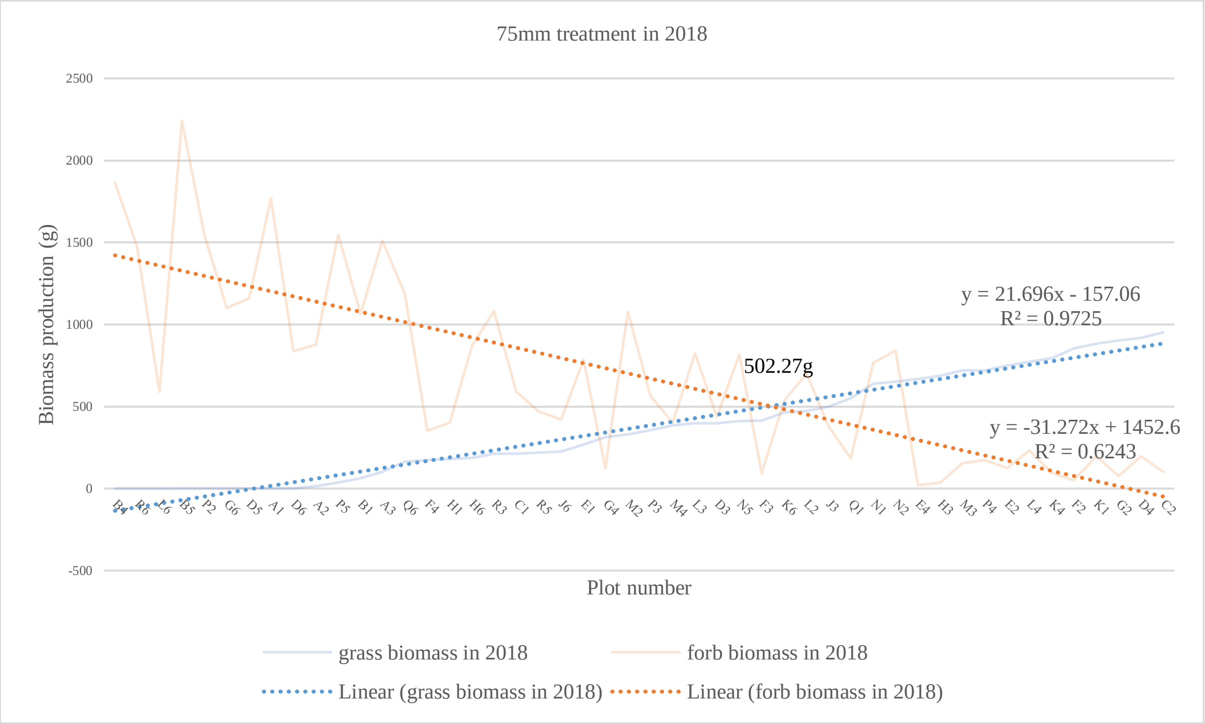


Figure 8.9 Grass biomass threshold within 75mm treatment on declines of forb biomass in 2018

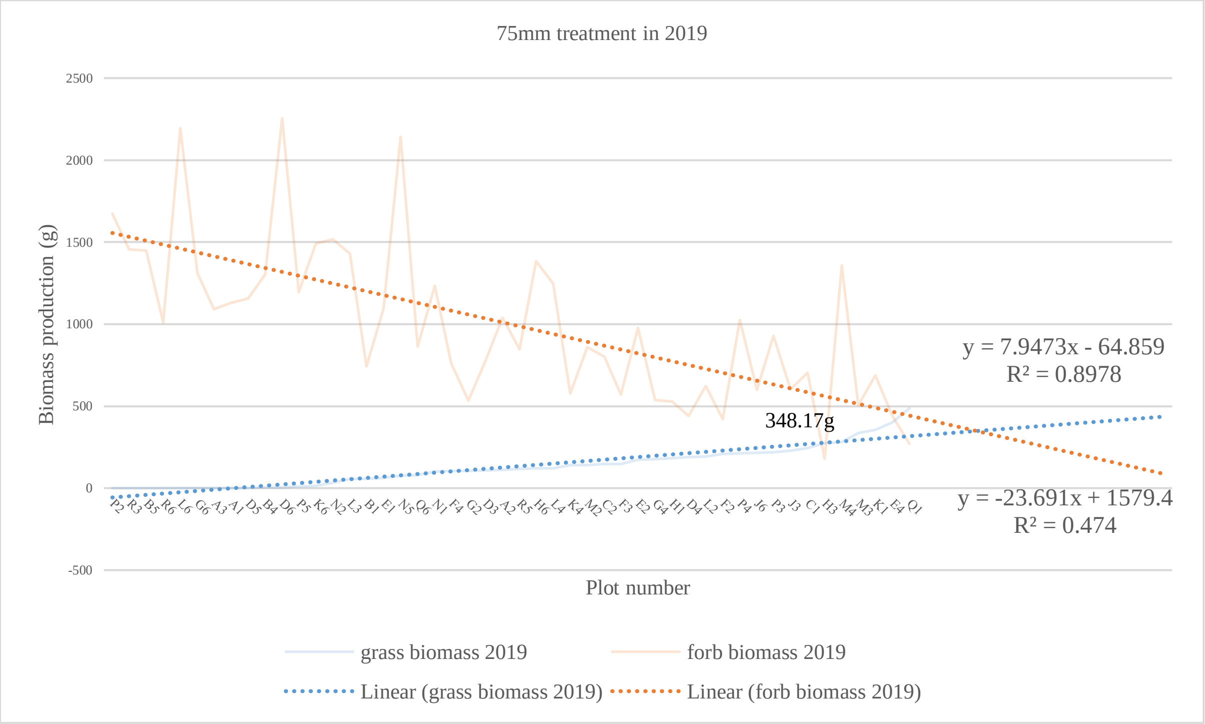


Figure 8.10 Grass biomass threshold within 75mm treatment on declines of forb biomass in 2019

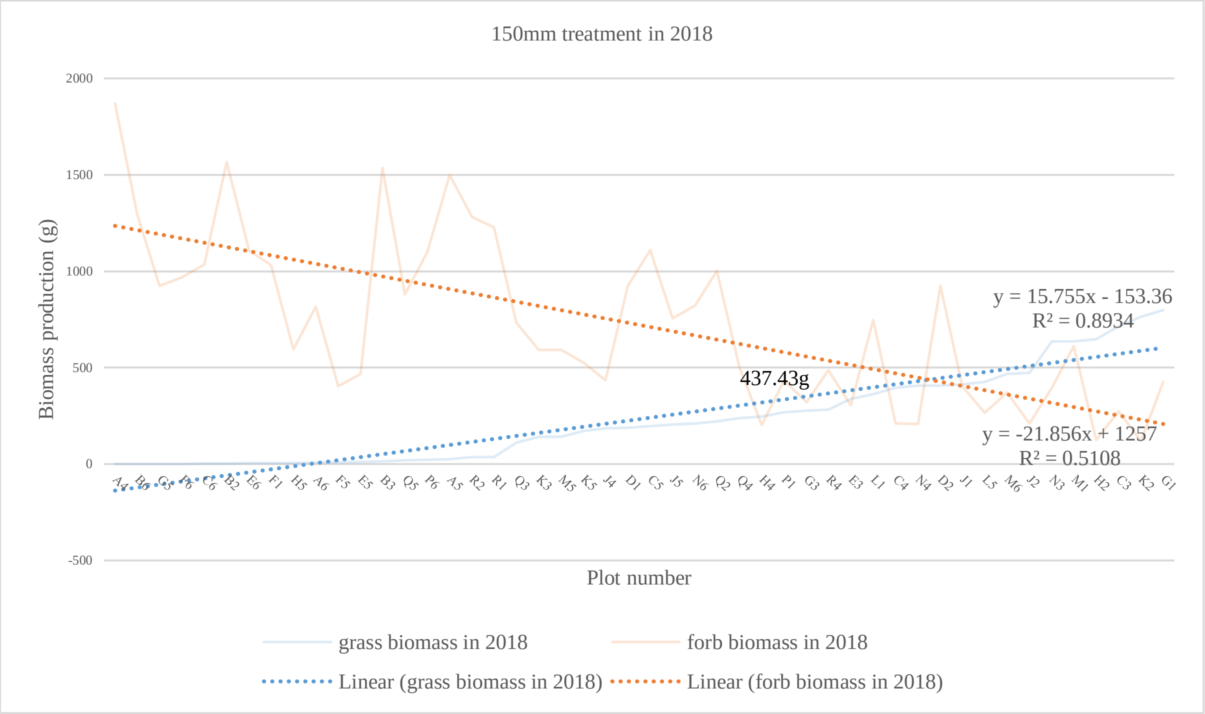


Figure 8.11 Grass biomass threshold within 150mm treatment on declines of forb biomass in 2018

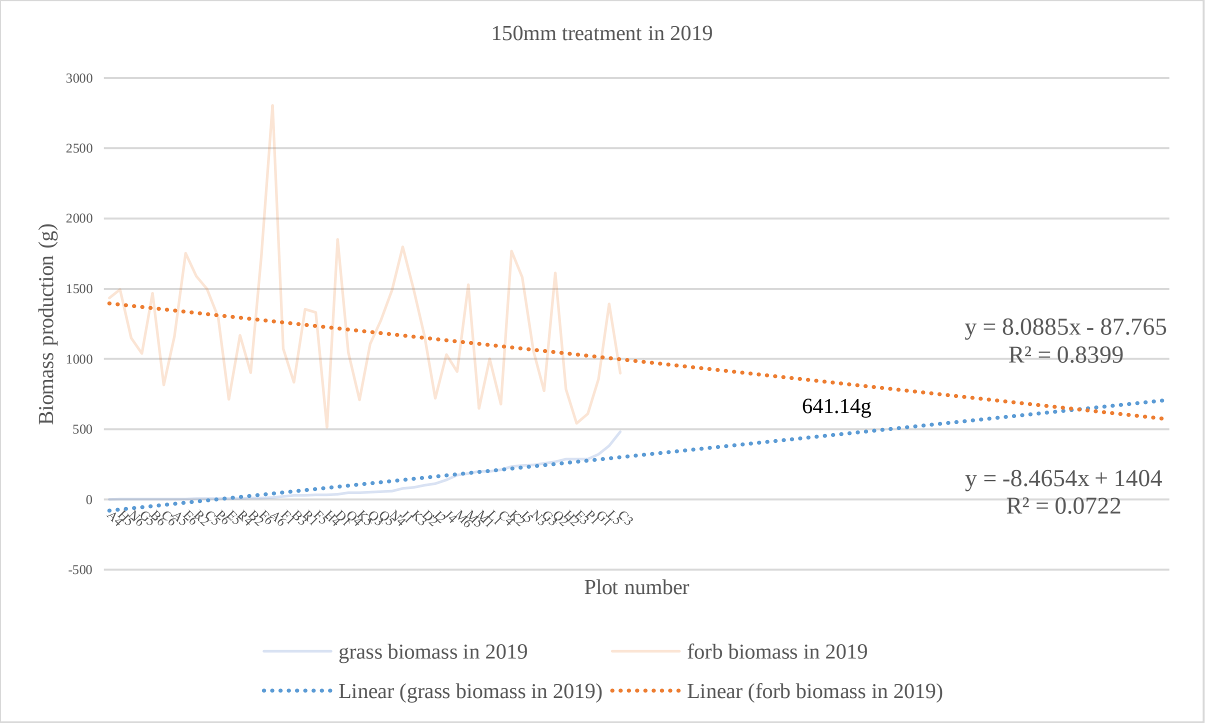


Figure 8.12 Grass biomass threshold within 150mm treatment on declines of forb biomass in 2019

8.1.4 Can a persistent forb rich and diverse meadow community be achieved in urban landscape through design?

In order to test the effect of the species competition in a relatively short term research period on relatively small plots, high sowing densities of grass, and medium and tall forbs soon led to a severe dominance and inevitable elimination. The scientific need to capture a minimum number of plants in each sampling frame (800 x 800mm in this study) to enable statistical analysis requires densities of large growing species well in excess to that which would be required in practice. Despite this initially high seedling density and dominance effects, forb diversity generally remained high to at least the third year.

From aesthetics point of view, the meadow community in this study was visually successful but could be improved as a model for the future landscape design. Twenty one out of 28 present forb species flowered from mid-April 2018 and 25 species in 2019 from early April. *Delphinium grandiflorum, Polemonium caeruleum* and *Pulsatilla vulgaris* flowered and added spring interests while *Achillea millefolium*, *Origanum vulgare* and *Sanguisorba officinalis* quickly recovered after the annual cutback in August and flowered until November to extend the landscape attractiveness. *Echinops sphaerocephalus* and *Deschampsia cespitosa* could be allowed to retain their structural dead shoots over winter as a design tool in landscape projects. Also, flowering, especially for rosette forbs, indicates a that the forbs had reached the certain size to provide nectar and pollen for native invertebrates and capacity to produce seed to establish new seedling recruits (Hitchmough and Wagner, 2013). However, although a forb dominated community was achieved through design, this was at the expense of the diversity of subordinate forbs. The key to enhance forb diversity and persistence is to maintain subordinate seedling richness and maximise their biomass (Grime, 1987; Mariotte *et al.*, 2013). These subordinates are under threats from both grass competition and competition from the most vigorous forbs. From 2019, *A. millefolium* principally took over the ‘grassy role’ to continue to suppress the subordinate forbs by its parallel mechanism of dense upright foliage to intercept light resources but due to its lateral rhizomes, greatly capacity for horizontal expansion. The evidence is that the lowest value of solar radiation level/ ambient appeared in the highest *A. millefolium* seedling and biomass present plots (1.19% in the treatment of ‘high sowing density’ x ‘seedling ratio of forb: grass of 9: 1’) according to “Sunscan” PAR measurement. Other flowering species, although present, due to their very small biomass were barely visible from a distance. This reduces the aesthetic values necessary to meet the needs of many urban landscapes. This study identified light competition played the major role to suppress subordinate forbs which was also commonly observed in other studies (e.g. Pywell *et al.*, 2003; Silvertown *et al.*, 2006; Hitchmough, Wagner and Ahmad, 2017). Edge effects appeared to be very important for the persistence of subordinate forbs, but because of the presence of a 200mm wide guard row around each plot, these subordinate forb populations were not captured in the data. In practice, dominance of the tallest most competitive species can be solved by reducing sowing density of both grass and dominant forbs of *A. millefolium* and *E. sphaerocephalus.* This would significantly increase flowering forb diversity and extend the visual effects in a longer term. Excluding colonising species such as *A. millefolium* all together would be an alternative strategy, although of course its role would then be assumed by other subordinates such as *Geranium pratense* with the next highest dominance potential. This study was, as with all PhD studies, time limited. In the longer term some of the subordinates would increase their biomass and become more visually impactful. Moreover, *E. sphaerocephalus* reached a height of about 2 metres on average from summer 2019 and its coverage was high (refer to Figure 8.13). In practice, the plant density of this species may need to be reduced in a gradient from a peri-urban to an ultra-urban context to meet the essential needs of sense of tidiness and safety in urban landscapes (Nassauer, 1995; Jorgensen, Hitchmough and Dunnett, 2007; Southon *et al.*, 2017). These issues are problematic in ecological experiments where resources do not permit the establishment and monitoring of larger, treatment plots.

Figure 8.13 *Echinops sphaerocephalus* reached about 2m in mid-July 2019. This made a feel of enclosed landscape when people walking on the path between the experimental plots where the density of *E. sphaerocephalus* was high

Sowing as a technique has potential to exploit the full range of plant density in a designed community to compete effectively with invading weeds and rapidly achieve visual effects from an early stage (Hitchmough and Fleur, 2006; Hitchmough, 2017b). It also enables designed meadow communities to be established at large scales particularly with a low initial resource input (Dunnett and Hitchmough, 2004). It is essential because a relatively small-scale meadow in a large greenspace, despite of forb dominance, may not effectively provide satisfactory interests and excitements for the public (Southon *et al.*, 2018). However, this approach requires more understandings to ecological process and species traits than conventional plantings from the outset. In term of sowing season (it was sown in summer because it was not possible to get the experimental field prepared for sowing in autumn 2016) despite this the expected seedling density was achieved by summer sowing, given additional irrigation (approximately every two days in absence of rain after sowing) and hessian stretched over the tops of experimental plots, which created approximately 50% more shades to keep the micro-temperature down and slow down drying out. High seedling mortality was observed particularly for *Deschampsia cespitosa* 'Barcampsia' after the hessian shades were removed (refer to Figure 8.14). Summer sowing potentially leads to low seedling emergence and higher mortality (Hitchmough, De La Fleur and Findlay, 2004; Hitchmough and Fleur, 2006). Species dynamic in designed communities may never appear exactly as expected due to the complexity of ecological process (Hitchmough, 2017a) but to reduce the sowing density of potential dominants is one of the most important tools for designers (Hitchmough and Wagner, 2013). *Achillea millefolium* seedlings were designed to be presence at densities from 1 to 26 seedlings per plot (according to the density/ratio treatment) in summer 2017, and actual numbers approximated to this in the experiment due to removal of excess seedlings. However, the rapid dominance of this species was unexpected even in the plots with the lowest *A. millefolium* seedling density. This shows that a small 'mistake' in sowing mix can lead to a dramatically different pattern and potentially a failed design. In practice, asymmetric competition can be driven by subtle advantage and this may be caused by various factors which is unlikely to be fully predicted (Hitchmough, 2017a). Thus, this study suggests the importance to identify and remove the potential dominant species seedlings as soon as possible as it becomes clear that a species is behaving more aggressively than anticipated. This is not of course possible in experiments designed to test hypotheses.

Figure 8.14 Some grass seedlings were observed to turn into brown as a sign of moisture stress in dry plots in August 2017, after the hessians were removed

8.1.5 To what extent, can design approaches enhance coexistence within meadow forb communities?

In this study, dominant forbs were at an advantage from the outset and soon suppressed the subordinates irrespective of the mix originally sown (Hitchmough and Wagner, 2013; Hitchmough, Wagner and Ahmad, 2017). The effect of ‘species composition’ generally appeared more important than the forb sowing and seedling densities for forb biomass diversity. Those design approaches merely changed the dominant biomass proportion between *Achillea millefolium*, *Echinops sphaerocephalus* and *Deschampsia cespitosa* without alleviating suppression of the subordinates. Forb seedling richness and biomass are unlikely to linearly increase with the sowing density or the seedling ratio of forb: grass since symmetric growth is something of an illusion. The possible way to enhance the subordinate biomass for a long time period is to allow more physical space for subordinate seedlings by lowering sowing density of the mix (Dickson and Busby, 2009). The low sowing density allows for space for subordinate seedlings and became more competitive to build more biomass. This was more evident in 2019 given lower levels of competition for light. The ‘A2’ plot accidentally had almost few grass seedlings at the beginning and a very low density of *Achillea millefolium* and *Echinops sphaerocephalus* (refer to Figure 8.15). However, the initial vegetative cover was extremely low. Some subordinate species that were considered to be unsuccessful in the communities on other plots in the study achieved a substantial biomass in this plot for example *Campanula punctata* (20.79g in 2018 and 8.47g in 2019 comparing with the mean value of 1.92g in 2018 and 1.04g in 2019), *Dracocephalum ruychiana* (6.13g in 2018 and 9.02g in 2019 comparing with the mean value of 0.86g in 2018 and 0.26g in 2019), and *Potentilla rupestris* (46.54g in 2018 and 121.00g in 2019 comparing with the mean value of 3.02g in 2018 and 4.20g in 2019). Thus, this study suggests that increasing seedling density for the subordinate species may be less effective than dropping the rate of dominants very low.

Figure 8.15 A2 plot which accidentally had almost no grass biomass and a very low density of dominant forbs in 2018. Subordinate forbs such as *Campanula puntata* and *Delphinium grandiflorum* which usually remain small grew well and flowered in summer (pictures taken on the 20th June 2018)

In terms of the design of seedling ratio of forb: grass, the highest forb subordinate biomass appeared at the ratio of 1: 1 in 2018. This is where *Achillea millefolium* seedlings were much lower comparing with the ratio of 9: 1 (forb: grass). This might suggest that in this scenario (especially the ‘tall grass’ and ‘150mm sand mulch’ treatments massively reduced grass competitiveness), *A. millefolium* were consistently competitive and more impactful to dominate than the grasses to the subordinate biomass within the experimental period. In 2019, the difference of subordinate biomass was reduced between the ‘ratio treatment’ with no statistical difference. This applies to the finding that the dominant canopy cover species were shading all plots irrespective of the designed species ratios; the starting point did affect numerical abundance of the species for several years but not their biomass (Hitchmough, Wagner and Ahmad, 2017). This again suggests a cost ineffectiveness to adding forb seeds to enhance subordinate forb biomass. However, the subordinate forb composition in 50% and 90% was more complex and made a greater visual impact in summer 2019. In respect of this, the higher initial seedling richness potentially creates more aesthetical values.

Despite grass coverage of ‘the short’ and ‘the tall grasses’ became similar in spring 2019, subordinate biomass was still significantly higher in the ‘tall grass’ treatment. Given the overall forb biomass appeared at a similar level between the two treatments (as mentioned above), this showed a better forb species coexistence. This suggests that a higher grass presence at the initial stage is more likely to eliminates the slow growing and shade intolerant species (Hitchmough and Fleur, 2006) which predominantly reduces forb species diversity in this study. This suggests that leaving grasses out of the sowing mix may be a useful strategy. Despite *Echinops sphaerocephalus* becoming the dominant species together with *A.*

millefolium, the ‘tall grass’ significantly reduced the biomass of *E. sphaerocephalus* where the grass coverage was substantial in spring 2019 (about 47% in the ‘tall grass x ratio of forb: grass of 1: 9’ and the 32% in the ‘tall grass x 75mm sand substrate’ treatment) whereas *A. millefolium* effectively escaped grass competition. This was predominantly because the taller grasses can effectively intercept more light incident on the predominant basal leaves of *E. sphaerocephalus,* and suppress its growth but *A. millefolium* had tall leafy stems above grass canopies. This shows a sign that the internal structure of the forb community and the dominance hierarchy was affected by height of grass, depending on the species architecture.

Although the ‘150mm sand mulch’ treatment reduced grass competition as well as slowed the growth of vigorous forbs, higher subordinate forb seedling numbers and lower biomass was found in the ‘150mm’treatment in both 2018 and 2019. Although the light stress to the subordinates is indirectly reduced with moisture stress, slow growing species are less able to optimise the utilisation of light resources due to limited water and nutrient uptake to support further photosynthesis and carbohydrate accumulation (Bannister, 1976; Grime, 2002). Also, the advantage of greater light access might be transient that the vigorous species overcome the stress sooner and still shade the subordinates. Del-Val and Crawley (2005) found that increasing herbivore pressure on a grass dominated community can have the opposite effect depending on the resource availability of the system. The author’s study argues the similar suggestion that imposing moisture stress to decrease dominants competition may also drive to different directions for subordinate biomass and this may depend on the key species characteristics of both dominant and subordinate.

Ranks the subordinate forb biomass in 2018 and 2019 from the highest to the lowest in each treatment combination (refer to Figure 8.16 and 8.17). Relatively higher moisture availability (i.e. the ‘75mm sand substrate’), reducing initial grass presence (i.e. the ‘tall grass’) and lower dominant forb density (i.e. the ‘high sowing density x ratio of forb: grass of 1: 1’ or ‘low sowing density x ratio of forb: grass of 9: 1’) were essential to enhance subordinate forb biomass to increase coexistence. On the contrast, grass competition and low sowing density of subordinate forbs (i.e. the ‘ratio of forb: grass of 1: 9’ and ‘short grass’ treatment) decreased subordinate biomass the most in 2018 while in 2019, moisture stress (i.e. the ‘150mm sand substrate’) appeared to be the major factor reducing subordinate biomass. Light competition is recognised as the major process for dominants to suppress subordinates (Grime, 2002; Köppler and Hitchmough, 2015) from the seedling stage. Lowering seedling density especially of vigorous species can increase the persistence of forb seedling richness, diversity and biomass abundance. Despite the fact that first year survival may represent the species composition in the longer term (Lauenroth and Adler, 2008), this study suggests the seedling biomass establishment in the first year may be more important than having more seedling survivals to enhance the subordinates performance in the longer term. To extend forb richness in the longer term requires some sort of disturbance interventions that were not part of this study. The author’s study argues that a ‘stable’ meadow is unrealistic, but a long-term forb rich meadow community might be achieved with appropriate management to extend the effect of original design.

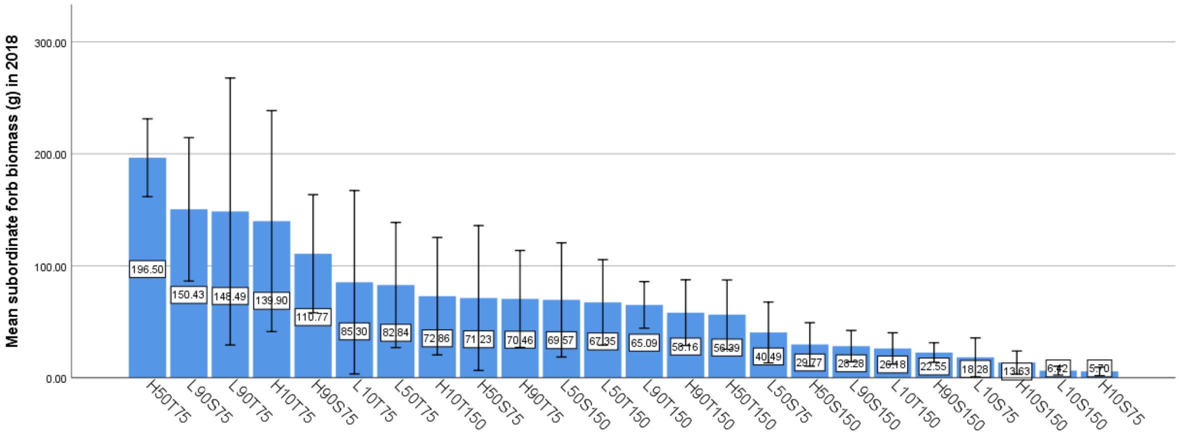


Figure 8.16 ranked subordinate biomass production (g) from the highest to the lowest treatment combination in 2018 (labels represent the combination of treatment for example H50T75 represents high sowing density x ratio of forb: grass of 1: 1 x tall grass x 75mm sand substrate)

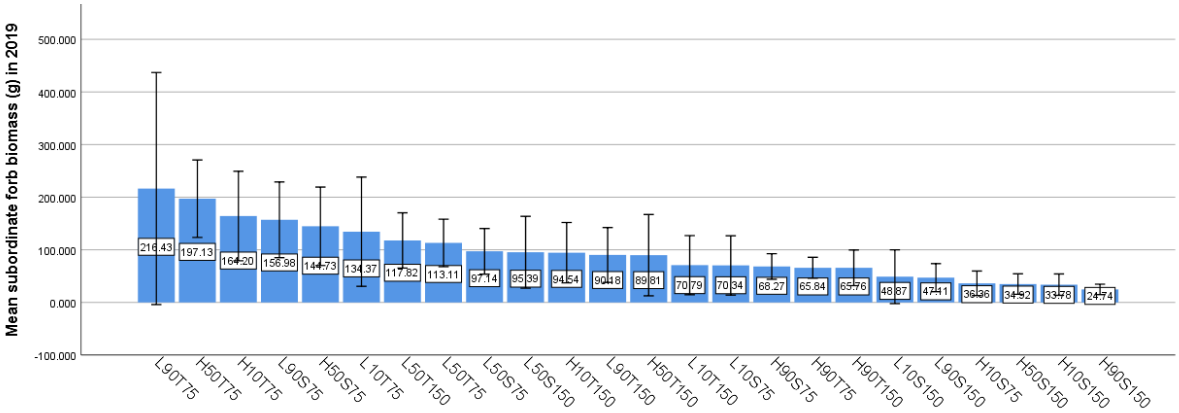


Figure 8.17 ranked subordinate biomass production (g) from the highest to the lowest treatment combination in 2019 (labels represent the combination of treatment for example L90T75 represents low sowing density x ratio of forb: grass of 9: 1 x tall grass x 75mm sand substrate)

8.1.6 What subordinate forbs tended to be most persistent in response to dominant's competition?

Enhanced abilities for light competition are the universal advantages for the subordinate forbs to persist with the dominant species. The strategy to access more light resources are various between different species. Primarily, the architectural forms of tall leafy stems (e.g. *Veronica longifolia*), clambering stems (e.g. *Galium verum*) and elongating petioles (e.g. *Geranium pratense*) are typically beneficial to increase the competitiveness of the subordinates (Hitchmough, 2009). Also, characteristics of early emerging in spring and quick recovery (e.g. *Potentilla rupestris*) after cutback support more biomass accumulation before the canopy is closed by the dominant (Weiner, 1990; Tremmel and Bazzaz, 1993). For most subordinate species the number of surviving seedlings can be increased by a higher sowing density or limiting competition from the dominants. However, far fewer subordinates could positively response to those treatments in terms of biomass production. Since the grass competition was identified as the predominant factor eliminating subordinate forbs, the biomass persistence may require developing taller foliage and utilising the light and space above the dense grass canopy. *Organum vulgare, Sanguisorba officinalis* and *Geranium pratense* appeared to be the most competitive subordinates to coexist with grass biomass. This largely depended on their combined advantages of both early development and beneficial architectural form for light competition. Shade tolerance can be an alternative strategy to allow species survival under dominants canopies (Gruntman *et al.*, 2017). Seedlings of *Polemonium caeruleum* and *Thalictrum aquilegiflorum* reduced sensitivity to dominant competition may because of relatively high shade tolerance according to Ellenberg's light indicator value (5). However, the design model that use shade tolerant species for the understorey layer may require far greater shade tolerant abilities, in Ellenberg terms ≤3 (Hitchmough, Paraskevopoulou and Dunnett, 2008; Hitchmough, Wagner and Ahmad, 2017).

The study also emphasises the importance of the first year establishment by providing more light and space resources to the subordinate species for a longer term biomass production (Lauenroth and Adler, 2008; Hitchmough, 2017a). Many subordinate species, especially those sensitive to grass competition such as *Campanula punctata* produced more biomass in the 'tall grass' treatment, despite the expansion of this grass experienced in the third growing season.

The overall pattern suggests that the taller canopy group appeared less sensitive than the medium and low canopy groups to the dominant competition. To some extent, height can determine competitiveness. This may largely be because the taller species mostly had tall leafy stems or upright growth form whereas the lower canopy species mostly have rosette-like or basal foliage. Also, the comparative rate of biomass production for the tall species were generally higher than the short species. Moreover, the characteristic such as late emergence prevented the taller canopy species from reaching an optimal height and these species were at greater risk of being eliminated (Hitchmough, 2009). Good examples were *Patrinia scabiosifolia* as well as the medium canopy species *Platycodon grandiflorus* and *Scutellaria baicalensis*. Ecological fitness was also essential to cope with stress in the relatively dry conditions. Despite having tall leafy stems, and high comparative rates of biomass production, *Aconitum carmichaelii* and *Angelica sylvestris* established poorly.

In this study, the native (shared) subordinate species appeared to be more competitive to coexist with grasses and dominant forbs. This might because the shared species (distributed across Western Europe and Inner Mongolia) had adaptiveness to a wider range of climate and soil conditions and this may enhance their ability to persist across the designed treatments (Pywell *et al.*, 2003). In this sense, the conclusion that native species tend to better compete with grasses does not seem to be compelling. Also, one of the most dominant forbs, *Echinops sphaerocephalus*, was a non-native species. Thus, this study still argues the key characteristics and strategies for light competition determines the competitiveness. The ecological fitness of individual species in relation to a certain condition often play a much more important role than the geographic origin, and non-native species often perform equally well or even better when they fit with a specific design or ecological need (Hitchmough and Wagner, 2013; Hitchmough, 2017a). Moreover, the question that remained unexplored in this study is that are selected Mongolian species at disadvantages from the outset in the community? A calibrating experiment was set up at Inner Mongolia Agricultural University to test the response of Mongolian forbs against the same species as in the UK experiment under their native climate conditions. However, no valid data was obtained due to their misinterpretation for the management work (i.e. desired species were massively weeded at the initial stage).

In terms of creating a species diverse community, many ecological studies emphases the value of having a diversity of traits of species including emerging time and growth rate etc. in a community (e.g. Pokorny *et al.*, 2004; Tilman, Reich and Knops, 2006; Partzsch and Bachmann, 2011). This makes perfect sense from a theoretical perspective assuming that there are always new potential immigrants from the outside, however it is not clear how this translates into communities in a landscape project which is to be established from a blank sheet, in which desired species composition is restricted to what is initially sown. Selecting species with similar key traits in relation to the site conditions can enhance the symmetric growth at the seedling stage and helps coexistence in the longer term. However, asymmetric patterns in species biomass is unavoidable with passage of time as it can be formed from accumulation of any subtle difference (Tremmel and Bazzaz, 1993; Hitchmough, 2017a).

8.2 Key take-away messages from the work

A series of hypotheses and research questions were tested to provide a connection and reflection between ecology and landscape architecture. This incorporated a lot of cross over factors and analyses and generated a piece of comprehensive but intense information in the final discussion. Thus, this section looks to clarify the key messages for both practice and future research for landscape architecture. The messages will be delivered at two levels of ‘community’ and ‘species’ and will focus on three main research questions that are, in short, do grasses inevitably outcompete forbs in the long term? Can mix design determine the community appearance in the long term? And which species better coexist with dominants?

COMMUNITY LEVEL		
Research focus	**Does grass biomass outcompete forbs in the long term?**	**Can design determine the community appearance in the long term?**
Key findings	- Forb dominance could be achieved and persist with a base layer of grass biomass in the long term through design in urban landscapes - This design approach involves low fertile substrate, relatively moisture stress tolerant forbs and moisture stress sensitive grass largely limiting the grass competition in the seedling stage - The factor of 'species' was more important 'the life form of forb or grass' for the longer term performance - However, grass biomass was still be hostile to most of the component forb species, mainly through closing down the canopy from spring onwards	- The design approaches affected the numbers of forb seedling in the long term but not the biomass - Ecological process (mainly competition) inevitably overrode initial design, and this effect could be significant from the short term. In this study, dominance effects were dramatic - Despite dominant forbs contributing to resistance to weed invasion and maximising forb productivity, these forbs exerted a negative effect on subordinate forbs in the same way that grass does - The key factors to enhance the subordinate forb biomass and the coexistence were 1) low grass and dominant forb density, 2) avoidance of severe stress (mainly moisture stress in this study) and 3) enhanced first year establishment
Recommendations for landscape practice	- Sensible design approaches can enhance the initial community establishment to achieve the benefits sooner. This also provides a good base for management work to extend the landscape values in the longer term - The author also argues that initial input for sown meadow community high sowing density of forb seeds and initial management is worthy. Despite the workload can be intensive, it is still relatively a small cost but effective comparing with other landscape works - Where forb biomass can effectively exclude invasion, grass can be left out to enhance subordinate biomass to some degree - Monitoring and interventions are suggested to take place as early as possible. Any minor difference within the sowing mix or environmental conditions can lead to a significant difference in biomass composition within a community, which potentially turns over the initial design within two years. To minimise the negative effect of dominance, identifying and decreasing the density of potential dominant species is the key to enhance community species richness - Overall, the most powerful design tool for a forb rich and diverse meadow community is the ratio between the component species especially dropping the dominants density to very low levels	

Recommendations for future research	- Long term studies (i.e. of at least 10 years) are needed. This study was relatively short term and other factors such as climate might significantly change competitive interactions in the longer term - Further studies on grasses and other dominant species biomass thresholds will be useful to inform any significant reduction to the richness of desired species	- As design may not be able to guarantee the persistence of sustainable urban meadow community without visionary management in practice, future studies will need to investigate the interactive effects between different design approaches and management strategies

SPECIES LEVEL	
Research focus	**Which forb species tend to better coexist with dominants?**
Key findings	- Despite native (shared) subordinate species appeared more competitive in coexisting with grasses and dominant forbs, the intrinsic adaptiveness to a wider range of climate and soil conditions may enhance their ability to persist across the designed treatments. Thus, geographical origins are far less important than the ecological fitness for species selection - Strategies for light competition determined the competitiveness of shade intolerant forbs to coexist with grass and dominant forbs - These strategies could include advantages in plant height and architecture (such as upright growth form and leafy foliage), early emergence, quick recovery and large seed size etc. The more of those characteristics a species had, the higher likelihood for it to establish - Also, any small difference in those characteristics between the species could lead to a highly asymmetric growth
Recommendations for landscape practice	- Selecting species with similar key traits (e.g. relative growth rate and architecture) can enhance the symmetric growth for a longer term - Different strategies to survive and compete are to utilise for different layers (e.g. shade tolerant forbs for the low forb layer)
Recommendations for future research	- Comparative studies for native and non-native species in relation to dominance effect will need further mirror experiments in both climate conditions - The practicability of classic ecological models such as CSR and Ellenberg Indicator Values will need to be further investigated for different conditions for urban meadows - The management history and processes in the original habitats will need to be incorporated and reflected in the future experiment design. This understanding will inspire the future proposal of management works

8.3 Conclusion

This study aimed to test the possibility of establishing a long-term forb dominance community coexisting with a base layer of grass biomass through design. This model is practically achievable, but a weed free soil and the initial establishment are essential. However, grass competition is still significant in reducing forb seedling survivals and biomass. The strategy to escape from grass competition predominantly depends on light competition. Abilities to project the foliage above grass canopies are essential for shade intolerant forbs. Although increasing the seedling ratio of forb: grass can achieve earlier forb dominance, this may increase the density of competitive forbs which then eliminate the subordinate forbs which represent most of the species diversity and reduce visual benefits of forb species richness by what is essentially ‘friendly fire’. Thus, it is important to minimise later asymmetric competition as much as possible at species selection and calculation of seedling sown density stage.

This study agrees with that a self-sustaining forb rich community is theoretically unrealistic (Bjørn, Weiner and Ørgaard, 2016). Community dynamics are not able to be fully predicted at the outset as it is affected by the factors at multiple dimensions. However, the design process can broadly determine the community development and create positive impacts over a relatively long term to satisfy the social needs for greenspace in urban landscapes. Also, this study argues that rather than perceiving planting communities as a product to be consumed with a diminishment in values with time period, these communities have potential with appropriate management to extend even increase both social and ecological values. Design is really important, but performance will depend on visionary management work in the long term.

8.4 Research limitation

Climate is the most uncontrollable factor in real life field experimentation. Therefore, annual climate difference may be a major force to change competition gradients. Also, extreme weathers may reduce experiment variables effects. For example, continuing rainstorm could make the designed sands layer soaked at a similar moisture level; and manual irrigation was applied in late spring and early summer in absence of rain to enhance the survival for valid data capture. Both situations might significantly reduce the treatment effects of moisture stress.

Despite selected species were mostly unpalatable and molluscicides (snail pellets) were sown in spring, the impact of herbivory could not be measured and had to be taken out as ecological factor.

The data collection period for seedling counting and biomass harvest usually took about three weeks to complete. The time period was not able to be reduced due to limited labour resources. The concern was the inconsistent pattern of community dynamic that varied from the beginning to the end of the work (i.e. late merging species tend to appear more seedlings, and early flowering species may start to dormant in the late data collecting plots).

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Appendices

Table A. Species characteristic table (over the page)

Species	**Categorisation in experiment**	**Origin**	**CSR**	**Relative Growth Rate (g/g/w)**	**Light (Ellenberg, 9 = full light; 1= deep shade)**	**Moisture (Ellenberg, 12 = permanently under water; 1 = extreme dryness)**	**Nitrogen (Ellenberg, 9 = extremely rich; 1 = extremely infertile)**	**Morphological charicteristic**	**Comparitive rate of biomass production (1 = low; 3 = high) ***	**Typical maximum leaf canopy height (mm)**	**Spring leaf phenology (1= earliest to emerge, 3 =latest)**	**Speed of recovery post cutting (1= earliest to emerge, 3 =latest)**	**Notes**
Anemone sylvestris	**Low canopy forb**	Shared	CS	NA	7	3	3	rosette/ foliage basal	1	<300	2	3	
Galium verum		Shared	CS	1.12	7	4	2	scrambling stems	1	300-450	2	2	
Potentilla rupestris		Shared	NA	NA	7	4	2	rosette/ foliage basal	1	<300	1.5	1.5	early growth and flowering
Pulsatilla vulgaris		Shared	NA	NA	7	3	3	rosette/ foliage basal	1	<300	2	3	early flowering and dormant
Veronica teucrium		Shared	C	NA	7	3	2	rosette/ foliage basal/some leafy stems	1	300-450	2	2	
Thermopsis lanceolata		Mongolian	NA	NA	NA	NA	NA	short leafy stems	1	300-450	1.5	3	big seeded
Dracocephalum rupestre		Mongolian	NA	NA	NA	NA	NA	rosette/ foliage basal	1	<300	1.5	3	
Dracocephalum ruyschiana		Mongolian	NA	NA	7	3	NA	short leafy stems	1	300-450	1.5	3	
Thalictrum petaloideum		Mongolian	CS	NA	NA	NA	NA	rosette/ foliage basal	1	<300	2	3	the least successful species
Veronica incana		Mongolian	NA	NA	NA	NA	NA	rosette/ foliage basal/some leafy stems	1	<300	2	3	the least successful species
Achillea millifolium	**Medium canopy forb**	Shared	C/ CSR	1.7	7	5	4	tall leafy stems	3	600-750	1	1	rhizomatous and creeping growth in sands; adapt wide range of habitat regarding literature
Stachys officinalis		Shared	S	NA	7	5	3	rosette/ foliage basal/some leafy stems	2	300-450	2	2	transplanted seedlings
Origanum vulgare		Shared	CSR	1.46	6	4	4	rosette/ foliage basal/some leafy stems	2	300-450	1.5	1	early emergence; quick recovery after cutback
Polemonium caeruleum		Shared	NA	NA	5	5	6	rosette/ foliage basal/some leafy stems	2	450-600	1.5	1.5	early emergence and flowering
Campanula punctata		Mongolian	NA	NA	NA	NA	NA	rosette/ foliage basal/some leafy stems	2	300-450	2	2.5	
Delphinium grandiflorum		Mongolian	NA	NA	NA	NA	NA	rosette/ foliage basal	2	300-450	1	1.5	early emergence and dormant
Kalimeris incisa		Mongolian	NA	NA	NA	NA	NA	tall leafy stems	2	450-600	2	1.5	
Platycodon grandiflorus		Mongolian	NA	NA	NA	NA	NA	tall leafy stems	2	450-600	3	3	late emergence
Scutellaria baicalensis		Mongolian	NA	NA	NA	NA	NA	rosette/ foliage basal/some leafy stems	1	600-750	3	3	late emergence
Echinops ritro	**Tall canopy forb**	Shared	C	NA	11	3	NA	rosette/ foliage basal/some leafy stems	2	<300	1	1 (but less tolerant to late cutback)	big seeded and early emergence
Geranium pratense		Shared	C	NA	7	6	7	dome of long elongating petioles	2	600-750	1	1	big seeded and early emergence
Sanguisorba officinalis		Shared	C/ CSR	NA	7	7	5	basal foliage with some leafy stems	2	450-600	2	1.5	
Thalictrum aquilegiifolium		Shared	CS	NA	5	8	8	basal foliage with some leafy stems	2	450-600	2	3	the least successful species
Veronica longifolia		Shared	NA	NA	7	8	7	tall leafy stems	2	600-750	2	2	'shoot thrust'
Aconitum carmichaelii		Mongolian	NA	NA	NA	NA	NA	tall leafy stems	2	>750	2	3	wet species
Angelica sylvestris		Mongolian	C/ CR	NA	7	8	5	tall leafy stems	3	>750	3	3	wet species
Echinops sphaerocephalus		Mongolian	NA	NA	8	4	7	basal foliage with some tall leafy stems	3	>750	1	1 (but less tolerant to late cutback)	big seeded and early emergence
Patrinia scabiosifolia		Mongolian	NA	NA	NA	NA	NA	tall leafy stems	1.5	600-750	3	3	late emergence, growth seems limited by low summer temperatures
Deschampsia cespitosa	**Grass**	Shared	C/ CSR	1.45	6	6	4				1.5		

* based on observation in cultivation in the UK

Table B. Forb species selection procedure

Initial Selection

Approximately **2,000** Mongolian species data from *E-flora of China* (2008), *Flowers of Mongolia* (2010), *Plants in Saihanba* (2010) and *Wild Flowers of Inner Mongolia* (2015).

↓

207 perennial species selected based on practical landscape value and potential environmental feasibility

↓

Habitat Study

112 species mainly from intermediate moisture habitat were selected and categorised into different canopy heights:

Low (≤300mm): 52 species

Medium (300 - 600mm): 40 species

Tall (≥ 600mm): 20 species

↓

Availability Check

52 species commercially available

↓

Feasibility Study

46 species selected based on tolerance of herbivory and populated into **heights x habitats cells**

Low (shared/ Inner Mongolian): 5/13

Medium (shared/ Inner Mongolian): 7/10

Tall (shared/ Inner Mongolian): 7/4

↓

Final Selection

29 species finally selected based on germination capacity from spring sowing to create equal cells of Heights x Habitats**.**

Low (shared/ Inner Mongolian): 5/ 5

Medium (shared/ Inner Mongolian): 5/ 5

Tall (shared/ Inner Mongolian): 5/ 4

Table C. Forb and grass seeds weighing regarding target numbers of emergence and estimated emergence % for designed sowing densities and ratio of forb: grass

(a) Low sowing density (500/m2)

500/m2

			Forbs: 90%		Forbs: 50%		Forbs: 10%	
Species	**Seed/g**	**Emergence %**	**Seedlings/m2**	**Seeds/g/m2**	**Seedlings/m2**	**Seeds/g/m2**	**Seedlings/m2**	**Seeds/g/m2**
Anemone sylvestris	2250	20.67	26	0.0559	14	0.0301	3	0.0065
Galium verum	3571	30	26	0.0243	14	0.0131	3	0.0028
Potentilla rupestris	4000	25	26	0.0260	14	0.0140	3	0.0030
Pulsatilla vulgaris	440	26.67	26	0.2216	14	0.1193	3	0.0256
Veronica teucrium	4234	30	26	0.0205	14	0.0110	3	0.0024
Thermopsis lanceolata	100	30	26	0.8667	14	0.4667	3	0.1000
Dracocephalum rupestre	400	30	26	0.2167	14	0.1167	3	0.0250
Dracocephalum ruyschiana	330	17.3	26	0.4554	14	0.2452	3	0.0525
Thalictrum petaloideum	780	25.6	26	0.1302	14	0.0701	3	0.0150
Veronica incana	10000	12	26	0.0217	14	0.0117	3	0.0025
Achillea millefolium	7500	30	13	0.0058	7	0.0031	1	0.0004
Campanula glomerata	7500	2.5	13	0.0693	7	0.0373	1	0.0053
Origanum vulgare	12000	5.33	13	0.0203	7	0.0109	1	0.0016
Polemonium caeruleum	1150	30.67	13	0.0369	7	0.0198	1	0.0028
Stachys officinalis	950	12.7	13	0.1077	7	0.0580	1	0.0083
Campanula punctata	12500	4	13	0.0260	7	0.0140	1	0.0020
Delphinium grandiflorum	1000	20	13	0.0650	7	0.0350	1	0.0050
Kalimeris incisa	980	30	13	0.0442	7	0.0238	1	0.0034
Platycodon grandiflorus	1000	33.33	13	0.0390	7	0.0210	1	0.0030
Scutellaria baicalensis	690	21.3	13	0.0885	7	0.0476	1	0.0068
Echinops ritro	83	21.33	6	0.3389	4	0.2259	1	0.0565
Geranium pratense	120	20	6	0.2500	4	0.1667	1	0.0417
Sanguisorba officinalis	400	24	6	0.0625	4	0.0417	1	0.0104
Thalictrum aquilegifolium	500	13.3	6	0.0902	4	0.0602	1	0.0150
Veronica longifolia	15244	7	6	0.0056	4	0.0037	1	0.0009
Aconitum carmichaelii	370	8	7	0.2365	5	0.1689	1	0.0338
Angelica sylvestris	460	20	7	0.0761	5	0.0543	1	0.0109
Echinops sphaerocephalus	75	20	8	0.5333	5	0.3333	2	0.1333
Patrinia scabiosifolia	1000	34	8	0.0235	5	0.0147	1	0.0029

500/m2			Grasses: 10%		Grasses: 50%		Grasses: 90%	
	Seed /g	Emergence %	Seedlings/ m2	Seeds/g/ m2	Seedlings/ m2	Seeds/g/ m2	Seedlings/ m2	Seeds/g/ m2
Deschampsia cespitposa 'Pixie Fountain'	4318	25	50	0.0463	250	0.2316	450	0.4169
Deschampsia cespitposa 'Barcampsia'	3448	10	50	0.1450	250	0.7251	450	1.3051

(b) High sowing density (1,000/ m2)

1,000/m2

			Forbs: 90%		**Forbs: 50%**		**Forbs: 10%**	
Species	**Seed/g**	**Emergence %**	**Seedlings/ m2**	**Seeds/g/ m2**	**Seedlings/ m2**	**Seeds/g/ m2**	**Seedlings/ m2**	**Seeds/g/ m2**
Anemone sylvestris	2250	20.67	52	0.1118	29	0.0624	6	0.0129
Galium verum	3571	30	52	0.0485	29	0.0271	6	0.0056
Potentilla rupestris	4000	25	52	0.0520	29	0.0290	6	0.0060
Pulsatilla vulgaris	440	26.67	52	0.4431	29	0.2471	6	0.0511
Veronica teucrium	4234	30	52	0.0409	29	0.0228	6	0.0047
Thermopsis lanceolata	100	30	52	1.7333	29	0.9667	6	0.2000
Dracocephalum rupestre	400	30	52	0.4333	29	0.2417	6	0.0500
Dracocephalum ruyschiana	330	17.3	52	0.9108	29	0.5080	6	0.1051
Thalictrum petaloideum	780	25.6	52	0.2604	29	0.1452	6	0.0300
Veronica incana	10000	12	52	0.0433	29	0.0242	6	0.0050
Achillea millefolium	7500	30	26	0.0116	14	0.0062	3	0.0013
Campanula glomerata	7500	2.5	26	0.1387	14	0.0747	3	0.0160
Origanum vulgare	12000	5.33	26	0.0407	14	0.0219	3	0.0047
Polemonium caeruleum	1150	30.67	26	0.0737	14	0.0397	3	0.0085
Stachys officinalis	950	12.7	26	0.2155	14	0.1160	3	0.0249
Campanula punctata	12500	4	26	0.0520	14	0.0280	3	0.0060
Delphinium grandiflorum	1000	20	26	0.1300	14	0.0700	3	0.0150
Kalimeris incisa	980	30	26	0.0884	14	0.0476	3	0.0102
Platycodon grandiflorus	1000	33.33	26	0.0780	14	0.0420	3	0.0090
Scutellaria baicalensis	690	21.3	26	0.1769	14	0.0953	3	0.0204
Echinops ritro	83	21.33	12	0.6778	7	0.3954	1	0.0565
Geranium pratense	120	20	12	0.5000	7	0.2917	1	0.0417
Sanguisorba officinalis	400	24	12	0.1250	7	0.0729	1	0.0104
Thalictrum aquilegifolium	500	13.3	12	0.1805	7	0.1053	1	0.0150
Veronica longifolia	15244	7	12	0.0112	7	0.0066	1	0.0009
Aconitum carmichaelii	370	8	15	0.5068	8	0.2703	1	0.0338
Angelica sylvestris	460	20	15	0.1630	9	0.0978	1	0.0109
Echinops sphaerocephalus	75	20	15	1.0000	9	0.6000	2	0.1333
Patrinia scabiosifolia	1000	34	15	0.0441	9	0.0265	1	0.0029

1,000/m2			Grasses: 10%		Grasses: 50%		Grasses: 90%	
	Seed /g	Emergence %	Seedlings/ m2	Seeds/g/ m2	Seedlings/ m2	Seeds/g/ m2	Seedlings/ m2	Seeds/g/ m2
Deschampsia cespitposa 'Pixie Fountain'	4318	25	100	0.0926	500	0.4631774	900	0.8337
Deschampsia cespitposa 'Barcampisa'	3448	10	100	0.2900	500	1.45011601	900	2.6102