Investigating The Planting Potential For Urban Rain Gardens: Plant Selection, Establishment and Performance

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Abstract

Rain gardens refer to planted shallow depressions widely adopted in urban areas to integrate vegetation and soil to mitigate the increasing urban stormwater issues, and are also perfect spots to adopt taxonomically diverse plantings to provide habitat values and aesthetics. However, few studies to date have successfully reflected the horticultural aspects and planting potential in rain gardens. This PhD is divided into three separate studies, which aim to characterise the success of a range of potential plants with different traits and different geographical origins in typical rain garden conditions and to access the relative hydrologic performance of different vegetation types to make informed planting decision, as well as to investigate the establishment of low-input in-situ sown vegetation in rain gardens.

The first study tested a range of potential native and non-native forbs and grasses in simulated rain garden cyclic flooding and extended drought. Results confirmed existing expectations with respect to which plants would be best suited to the bottoms, slopes and margins of periodically-inundated rain gardens.

In the second study, experimental rain gardens planted with taxonomically diverse plantings composed of forb-rich perennials, mown grasses and bare soils were tested with artificial rainfall. The forb-rich perennial mixes featuring greater species richness and structural diversity consistently provided the best hydrologic performances, and can therefore be recommended for use in rain gardens.

In the third study, in-situ sown forb-rich plantings were created in practical rain gardens with the involvement of two low-impact weed control measures including the use of felt mats and mulching. Mulching shows significant effectiveness on weed control, whereas no valid conclusion could be drawn on the effectiveness of felt mats due to the contamination of the potentially weedy compost mulching. The ‘dry-wet’ moisture gradient in rain garden depression was determined to significantly influence the establishment of sown plantings.

Acknowledgement

This thesis is for people who share a love of gardening and ecological landscape design. We are now in an era where the environment changes greatly challenge the survival and growth of cities. Our cities are vulnerable to the severe stormwater issues and damaged habitats resulting from the rapid urbanisation and climate change. The urban landscapes become progressively lifeless places where plant diversity is missing. I chose the research topic for exploring the planting potential for urban rain gardens because of my love of water in all its natural forms and the natural wildflower wonders across the world, as well as my dream and passion to create rain gardens with dynamic and exciting flowering appeals in cities to minimise stormwater issues and to bring back the colourful mother nature.

During past four years, I had a truly great and exciting PhD journey at University of Sheffield. I remember so vividly the inspirations and passion with which I learned in Landscape department and created in the fields in Green Estate Ltd., Sheffield. The amazing journey has changed my life, and I feel very grateful for all the happiness that the study has brought to me and for all the people who have made invaluable contributions to my PhD study.

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Table of contents

List of Tables

Page

Chapter 4

Chapter 5

List of Figures

Page

Chapter 2

Chapter 3

Chapter 4

<u>Chapter 5</u>

Chapter 6

Chapter 1: Introduction

Rapid urbanisation and climate change drive adverse changes in urban environment and ecological processes, which result in serious issues such as air and water pollutions, increasing water consumption and greenhouse gas concentration, fragmentation of habitats and extreme weather disasters (e.g. flooding and drought), which significantly threaten the sustainability of cities and civilisation (Grimm et al., 2008). Water, as the most precious resource to society, is governed by hydrological cycle including the different ecological processes of precipitation, infiltration and evapotranspiration (Woelfle-Erskine & Uncapher, 2012). However, increasing impervious surfaces due to urban development modifies the land cover permeability to reduce infiltration and evapotranspiration, and this not only increases the amount of urban runoff (Poff et al., 2006), but also reduces the groundwater recharge to threaten water supply in urban areas (Lee et al., 2008). Current climate change predictions suggest extreme weather events such as intense storms and extended droughts will increase in urban areas (Meehl et al., 2004). The traditional urban stormwater management approaches constrained by piping stormwater away to the nearest water bodies are increasingly untenable to cope with the increasing urban runoff and associated pollutions, and do not take an opportunistic stance to use stormwater as a valuable resource (Kenway & Lant, 2012). Therefore, large urban areas are subject to urban flooding issues, drought orders, water shortages, and sewage matters that are harmful to the urban environment (Woelfle-Erskine & Uncapher, 2012;

Scholz, 2004).

Vegetation and soils are two vital factors that not only improve landscape hydrology, but also provide various ecosystems services such as climate regulation, pollution treatment, biodiversity conservation and aesthetics (Gill et al. 2007; Millennium Ecosystem Assessment, 2005). Vegetation and soils are thus largely used as ecological engineers to mitigate the adverse effects resulting from urbanisation and climate change in urban areas. Green infrastructure that uses landscape systems to restore the ecosystems in urban areas is widely adopted on a global scale to maximise the vegetation-soil based ecosystem services to sustain the growth of cities (Tzoulas et al., 2007). To respond to the issues resulting from the damaged urban water cycle, sustainable stormwater management components such as rain gardens, bioswales, green roofs and retention basins, have been integrated into the urban green infrastructure in the last two decades to counter the negative impacts of urbanisation and climate change on affected urban stormwater processes (Brown et al., 2009; Farrelly & Brown, 2011). These features can be flexibly retrofitted in existing urban developments to replace the impervious surfaces, and work as landscape sponges to capture, store, infiltrate and clean stormwater runoff on-site (Davis, 2005; Dietz, 2007). The sustainable stormwater management components can also add various ecosystem services derived from vegetation and soils to mitigate the expense of green spaces and the associated issues such as the loss of habitats and biodiversity, air and water pollution, as well as the lack of visual amenities, etc. (Howe & Mitchell, 2012;

Trowsdale & Simcock, 2011).

Rain gardens are small-scale land-based sustainable stormwater management components, and are the most popularly adapted among all the components in residential, commercial and municipal developments (Steiner & Doom, 2012). Rain gardens comprise shallow depression to catch and treat stormwater runoff and rely on natural precipitation as irrigation input (Dunnett & Clayden, 2007). Therefore, there is a typical cyclic flooding condition repeating between inundations in wet seasons and draining in dry seasons, as well as the potential extended drought in rain gardens (Dunnett & Clayden, 2007). There is also a 'wet-dry' moisture gradient throughout the typical depression structure in rain gardens (Dunnett & Clayden, 2007). Rain gardens are perfect public spots to demonstrate low-impact vegetation with great appeals that are attractive to city inhabitants and greatly contribute to local biodiversity (Dunnett & Clayden, 2007; Steiner & Domm, 2012).

Currently, rain gardens tend to be universally adopted as one of the most widely retrofitted practices in the urban green infrastructure and the sustainable urban stormwater management systems. Rain garden couple ecological systems with anthropogenic living environments, so that a holistic consideration of their ecology and aesthetics is required with the basic function of stormwater management (Pickett & Cadenasso, 2008). Vegetation ecosystem services are vital to the success of rain gardens. However, planting design is often underestimated in every stage of the implementation of sustainable stormwater management components. The general lack of consideration of vegetation diversity and aesthetics is one of the main concerns in contemporary urban green infrastructure as well as in sustainable stormwater management facilities such as rain gardens.

Plant diversity is vital to the success of the rain garden plantings, which not only governs the stability and aesthetics of vegetation over time, but also contributes to the biodiversity conservation and stormwater runoff treatment (Dunnett et al., 2008; Steiner & Doom, 2012). It is therefore recommended that vegetation with great plant diversity such as the taxonomically diverse wildflower meadows and prairies be adopted in rain gardens (Dunnett & Clayden, 2007). Nevertheless, plant diversity in sustainable stormwater management components is restricted due to the limited knowledge of professionals, while plant options are imposed to be native only where ecologists are involved. Thus, conventional urban vegetation of monoculture or plant communities with rather low species richness (e.g. mown grasses) that only provide rather limited habitat and aesthetic values are commonly seen in urban rain gardens (Dunnett & Clayden, 2007). There is therefore a constant requirement to expand the plant selection for improving the plant diversity in rain gardens.

Proper plant selection and implementation are invaluable to the success of rain garden vegetation for providing the basic stormwater runoff management and other ecosystem services (Johnston 2011; Shaw & Schmidt, 2003). However, there are rather limited researches providing data-based conclusions on the growth performances of preferred plants in typical cyclic flooding or the potential drought conditions in rain gardens (Dylewski et al., 2011). The understanding of the mechanistic interaction between vegetation and the hydrologic cycle in rain gardens is not yet clearly addressed (Johnston, 2011). Furthermore, rather limited conclusions on the success of the stormwater management of the recommended taxonomically diverse plantings in rain gardens could be drawn from previous studies. All the concerns become apparent and greatly challenge the professionals to make informed planting decisions for rain gardens. Therefore, quantifying the interactions between plants and typical rain garden conditions, as well as determining the role of vegetation in stormwater management especially of the potential differences of runoff treatment between the taxonomically diverse plantings and the conventional less diverse vegetation (e.g. mown grasses) is a foundation in the improvement of rain garden plantings.

In practice, vegetation establishment in sustainable stormwater management components such as rain gardens should be low-impact and cost-effective. Dunnett and Clayden (2007), Hitchmough and Wagner (2013) recommended sowing seed mixes in-situ as the alternative vegetation technique in rain gardens for reducing the inputs in planting implementation and maintenance. However, urban soils often tend to be productive and weedy (Hitchmough et al., 2008), so that the success of sown plantings in rain gardens may be greatly challenged by weed competition. In practice, the intense weeding in rain gardens could be costly, while the use of herbicides in rain gardens might cause severe chemical damage to downstream aquatic environments, especially when rain gardens are directly connected with public drainage systems (Yang et al., 2013). Therefore, simple and less interventionist weeding approaches are needed for the sown plantings in rain gardens. Furthermore, few reports reflect the success of the sown taxonomically diverse plantings throughout the particular moisture gradient in the depression structure in rain gardens (e.g. the dry and hardly waterlogged margin, moderate moist slope and constantly wet bottom. Dunnett & Clayden, 2007), which also leaves a great research gap and challenge in practice.

1.1 Goal and specific objectives

The goal of this PhD study is to provide a link between the planting and engineering and ecology aspects, which often seems to be missing in the rain garden implementation. This thesis considers the range of plants with different traits and geographical origins that are assumed well-suited to the typical moisture conditions (e.g. cyclic flooding and the gradient of moisture levels) in rain gardens, and also to provide solid data on their effectiveness. The effectiveness of two simple weed control methods (the use of felt mat and mulch) for sown taxonomically diverse plantings in the typical rain garden conditions and the interaction between sown forb-rich seed mixes and the typical conditions in practical rain gardens are also explored. The overall goal will be achieved with the following three objectives:

1. Quantify the effect of typical cyclic flooding and drought in rain gardens on the growth and survival of different plants and corresponding plant types to give informed decisions on extending the plant options for rain gardens;

2. Investigate the quantitative evidence of the change in stormwater runoff quantity in rain gardens by introducing the taxonomically diverse plantings compared to the traditional urban vegetation type such as mown grasses;

3. Evaluate the effectiveness of potential less interventionist weeding approaches on sown taxonomically diverse plantings in typical rain garden conditions, and the interaction of the seed mixes with the 'wet-dry' moisture distribution in rain gardens.

1.2 Thesis overview

Chapter 2 is a review of literature concerning the importance and contemporary issues of plantings in rain gardens. It also includes the reviews on (a) the overall background knowledge of urban stormwater issues related to the urbanisation and climate change and the role of vegetation, soils and green infrastructure in delivering ecosystem services, (b) importance, components and current developments of sustainable stormwater management, and (c) the overview of rain garden design aspects, plantings in rain gardens and the contemporary issues within the rain garden plantings, as well as the success of rain gardens in stormwater treatment, pollution control and ecology. Chapter 3 includes material on the selection of suitable plant species that are intended for use in typical rain garden conditions (i.e. cyclic flooding and moisture gradients), focused literature review, discussions with key findings. Chapter 4 generates data on the runoff retention (i.e. the total reduction in runoff) and detention (i.e. the temporary storage of runoff) among the proposed taxonomically diverse forb-rich plantings, mown grasses and bare soils in model rain gardens. It also provides a focused literature review and discussions on experimental observations. Chapter 5 investigates the potential of compost mulching depth and felt moisture mat as two key weed control measures for the in-situ sown plantings in rain gardens and the interaction between seed mixes and the moisture gradient in rain garden settings, while adding the focused literature review and discussions of the key findings.

Chapter 2: Literature Review

This chapter will review literature on the influences of urbanisation and climate change on urban environments. The general ecosystem services of vegetation, soils, and the green infrastructure combining urban vegetation and landscape for mitigating the negative impacts resulting from urbanisation and climate change, are presented. More specific details of the urban stormwater and all the associated issues caused by urbanisation and climate change are discussed. The effects of the sustainable stormwater management and on urban stormwater, an overview of all the facilities as well as the current application developments of sustainable stormwater management are presented. Specific examples and the relevant literature that relates specifically to design aspects and plantings of the most effective and widely-adapted terrestrial stormwater management facility, rain gardens, are discussed. The research gaps in the contemporary plantings in rain gardens, as well as the need for more research into plant selection and the establishment of sown plantings in typical conditions of rain gardens will be critically evaluated. Finally, hydrological performance of rain gardens and rain garden vegetation is critically reviewed, while the overview of the environmental values including pollution control and ecology conservation in rain gardens is also presented.

2.1 The main challenges facing cities including urbanisation and climate change scenarios

Urban areas are hot spots that drive environmental change, with increasing challenges for societies and ecosystems (Grimm et al., 2008). Urbanisation and climate change are the two dominant issues transforming the relationship between cities and the global environment, which tangle ecological and social conditions in urban areas and constitute major threats to the survival and sustainability of cities and well-being of urban inhabitants.

Rapid urbanisation drives environmental changes by altering land surface properties, hydrological systems and ecosystems (Grimm et al., 2008). Urban expansion is one of the primary drivers leading to the significant loss and alteration of green spaces and natural habitats, which ultimately result in loss of biodiversity and plant and animal species extinction in urban areas (McDonald et al., 2008; McKinney, 2002; Zhou & Wang, 2011). The rapid growth of human population, industries, vehicles and fossil fuel uses, increases water consumption and carbon dioxide (CO_2) concentration in cities, resulting in serious water stress and greenhouse gas-induced warming that challenge the global survival and sustainability (Grimmond, 2007). Furthermore, the presence of substantial air, water and noise pollutions in the built environment greatly damage the health and well-being of city inhabitants (Vlahov & Galea, 2002), while the loss of green spaces and natural environments are found to negatively affect the mental health of urban residents (Jackson, 2003; Van den Berg, et al., 2010).

Global warming, and marked climate fluctuation are two dominant climate change scenarios reported to cause observed damage in physical and biological systems in the urban context (Menzel, 2006) and terrestrial ecosystems in particular (Walther et al., 2002). The reported great temperature shifting and heat island effects resulting from the increasing greenhouse gas emissions from energy consumption, vehicle miles travelled, and fossil fuel uses in urban areas is expected to significantly influence the land-atmosphere interaction (i.e. the effects of soil moisture on precipitation) causing climate variability and the potential migration of climate zones (Seneviratne et al., 2006; Koster et al., 2004). For example, the current estimates suggest a further climate shift that would cause heavily modified thermal climates in many British regions (Boardman, 1990; ICE, 2012; Anon, 2009), such as the emergence of a more Mediterranean climate and a decline in summer rainfall (Hulme et al., 2002; Broadmeadow et al., 2005). The heat island process combined with global warming could not only result in a conspicuous growth of society's demand for water (Niemczynowicz, 1999), but also lead to rises in mortality in cities by damaging human health such as causing respiratory diseases (Haines, 1991). Heat island effect could greatly threat wildlife by increasing the degradation and loss of habitats, spreading diseases and pests.

Warming can certainly influence the abundance and distribution of species, and thereby affecting biodiversity in urban ecosystems (Petchey et al., 1999). Warmer temperature is reported to increase photosynthates in plants governed by solar energy, which lead to greater primary productivity from vegetation (Yvon-Durocher et al., 2010). Warming may therefore cause the migration and adaptation for more plant species from outside the region, which may possibly lead to greater biodiversity in general in urban ecosystems (Walther, 2003). However, warming may also increase the risk of introducing highly aggressive exotic invasiveness to decrease indigenous plant diversity (Thuiller, 2005; Wilby & Perry, 2006), though it may be less of a problem in the UK due to the ‘island’ ecological effect that English channel may act as a barrier to terrestrial species movement towards north as temperature rise (Bond et al., 2005).

Climate models have predicted an increased frequency and intensity of climate fluctuation in many regions of the world, which would result in increasing unpredictable weather events of extreme impact and profound repercussions (Meehl et al., 2004), such as the 2012 Hurricane Sandy in Eastern United States, severe drought-flood fluctuation in 2012 summer in Britain. The increasing extreme weather events resulting from more intensive climate fluctuation exacerbate the already damaged urban living environments (Smith, 2012) and threats to the ecosystem structure and function by affecting the distribution, abundance and individual fitness of plant and animal species and their population dynamics (Parmesan et al., 2000;

Walther et al., 2002). The increasing extreme weather events also result in steep challenges toward urban safety and sustainability. For instance, two-thirds of the world's population would live in high water-stressed regions by 2025 due to increasing droughts (WWAP, 2015). Extreme flooding events have caused a total cost in excess of US$ 1.3 trillion (63% of all costs due to damage of weather and climate disasters) and affected 4.2 billion people (95% of all population affected by weather and climate disasters) since 1992 (WWAP, 2015).

2.2 The ecosystem services of urban vegetation and soils in general and the green infrastructure implications in cities

In the face of these major challenges toward urban sustainability, efficacious solutions that promise or, ideally, are proven to be sustainable are urgently needed to mitigate the negative effects resulting from urbanisation and climate change. Urban vegetation and soils as two key factors governing a variety of important ecosystem services (i.e. the benefits provided from ecosystems to human beings. Bolund & Hunhammar, 1999) that sustain the resilience of cities (Bonan, 2008; Gill et al. 2007) has been long recognised, and thus have become ecological tools for mitigating the effects of urbanisation and climate change. Their ecosystem services include, but are not limited to: regulation of local climate, hydrology modification, sewage treatment, local biodiversity support and aesthetic and recreational values (Daily, 1997; Gill et al. 2007; Millennium Ecosystem Assessment, 2005; Mooney et al., 2009).

The role of vegetation and soils and their interaction on urban environments remains a frontier in ecological research (Pataki et al., 2011). The delivery of ecosystem services in urban areas is vulnerable to the land use changes, where the increasing lifeless built environments significantly reduce habitats and damage the functional diversity (i.e. the value, distribution and abundance) of biological communities (Díaz et al., 2007). Plants are the providers of the key ecosystem services in urban ecosystems including fuelling the entire terrestrial food chain and providing nest niches, host species and shelters to sustain the population of local species and biodiversity (Daily, 1997; Tilman & Downing, 1994; Wäckers, 2005), sustaining the soil fertility and earth productivity (Bignal & McCracken, 2000, Gibon, 2005). Soils provide the natural stocks of carbon, nutrients and water to support the productivity of plants and the soil biodiversity (including the soil microorganisms and macro invertebrate communities) (Lavelle et al., 2014; Schröter et al., 2005). Vegetation and soils are therefore the most important factors that support the community structure and functions in urban ecosystems.

Recent studies of the plants and soils in urban climate regulation identified two key ecosystem services. These are evaporative cooling from vegetation and soils (Onmura, 2001; Robitu, 2006), and the carbon sink service regulating the fluxes of the greenhouse gas emissions by vegetation and soil organisms (Daily, 1997; Scurlock & Hall, 1998). These vital ecosystem services could mitigate urban warming and the associated environmental degradation as noted previously. Replacing the impermeable land covers with vegetation and soils could greatly increase the infiltration and evapotranspiration in built environments, so that the urban hydrology can be restored (Waring & Landsberg, 2011; Johnston, 2011). Vegetation could greatly contribute to the removal of air pollution and noise control to improve the quality of urban living environments (Beckett et al., 1998; Fang & Ling, 2005; Nowak et al., 2006). Vegetation and soils are also widely used for treating the concentrations of a wide range of heavy metals and nutrients that are mobilised and transported by urban surface water (Alloway & Jackson, 1991; Raskin et al., 1997; Lee et al., 2010; Yang et al., 2013). Furthermore, urban vegetation not only greatly adds aesthetic values to be attractive to people (Kingsbury, 2004), but also positively influence human physiological and emotional states and thus promote the wellbeing of urban inhabitants (Ulrich, 1986).

To utilise the large number of important vegetation-soil based ecosystem services for maintaining ecosystem health, as well as providing multiple human-social benefits, the concept of ‘green infrastructure (GI)’ has been promoted worldwide (Tzoulas et al., 2007). GI refers to the networks that consist of the natural, semi-natural and synthetic ecosystems in urbanised areas (Tzoulas et al., 2007), which include the urban open green spaces that provide the ecological habitats for supporting ecological processes and wildlife (e.g. nature reserves, forest preserves, wetlands and parks, etc.) and links that provide corridors to connect the isolated green spaces and enable green infrastructure networks to work (e.g. greenways and greenbelts, etc.) (Lovell & Taylor, 2013; Weber et al., 2006). The concept of GI emphasises the restoration and creation of urban green spaces (Rudlin & Falk, 1999), interconnections between habitats within the green spaces (van der Ryn & Cowan, 1996) and their multifunctional role such as aesthetics and economical sustainability (Sandström, 2002; Walmsley, 2006).

The delivery of the main ecosystem services of GI such as the climate regulation, hydrology modification, urban biodiversity conservation and aesthetic and recreational values are largely driven by vegetation and soils. Planting design and soil amendments have been introduced to upgrade the contemporary GI, not only following the instructions of ecology science to enhance their structures and longevities, maximise their productivities and resilience to environmental changes to sustain urban ecosystems and biodiversity over time (Hunter, 2011), but also installed from the perspectives of urban design depending on human-social values and desires to provide public amenities and increase the property values (Benedict & McMahon, 2012; Hopkins & Goodwin, 2011). For instance, ecologists often suggest the use of native plants or plant species that hail from habitats with similar environmental conditions (e.g. temperature, moisture, sun, shade and wind, etc.) to ensure their adaptation to local conditions (Beck, 2013). However, the living systems in urban green infrastructure are heavily influenced by human activities and highly managed, where the lack of plant diversity adversely alter the urban biodiversity as well as the longevity and amenity of green infrastructure (Beck, 2013; Grimm et al., 2008). Thus, it is also proposed to increase plant species richness to keep functions and aesthetics of plant communities stable and support related wildlife population over time (Beck, 2013). Soil amendments are also encouraged by engineers for reversing soil compaction to increase stormwater drainage and groundwater recharge, which could also enhance the air movement in soils to encourage plant growth and decimate populations of beneficial soil organisms (Beck, 2013). Urban designers and landscape architects may seek more visually dramatic scenes using beautiful flowering plants and those of preferred structures to sustain the landscape effects of GI over time or planting vegetation in certain forms such as matrixes, borders or belts for pleasing visual appeal (Oudolf & Kingsbury, 2013), mixing or replacing existing soils with a variety of substrates to promote the growth of ornamental plants or purely for visual amenity.

2.3 Urban stromwater and associated issues

Water plays a significant role in everyday life and society relies on the supply and security of reliable water and resilience to water excess. Water appears in cities as (a) potable and importable water for daily use, (b) precipitation and surface runoffs, (c) natural water bodies (e.g. rivers, lakes, etc.) and artificial water bodies (e.g. fountains, streams and ponds, etc.), (d) groundwater and (e) greywater and waste water managed by urban sewage systems (Novotny & Brown, 2007; Anton, 1993). The complete urban water cycle includes precipitation, evapotranspiration, infiltration and surface runoff (Beck 2013). Fig. 2.1 shows some of the different water forms with both natural processes and alternations resulting from developments.

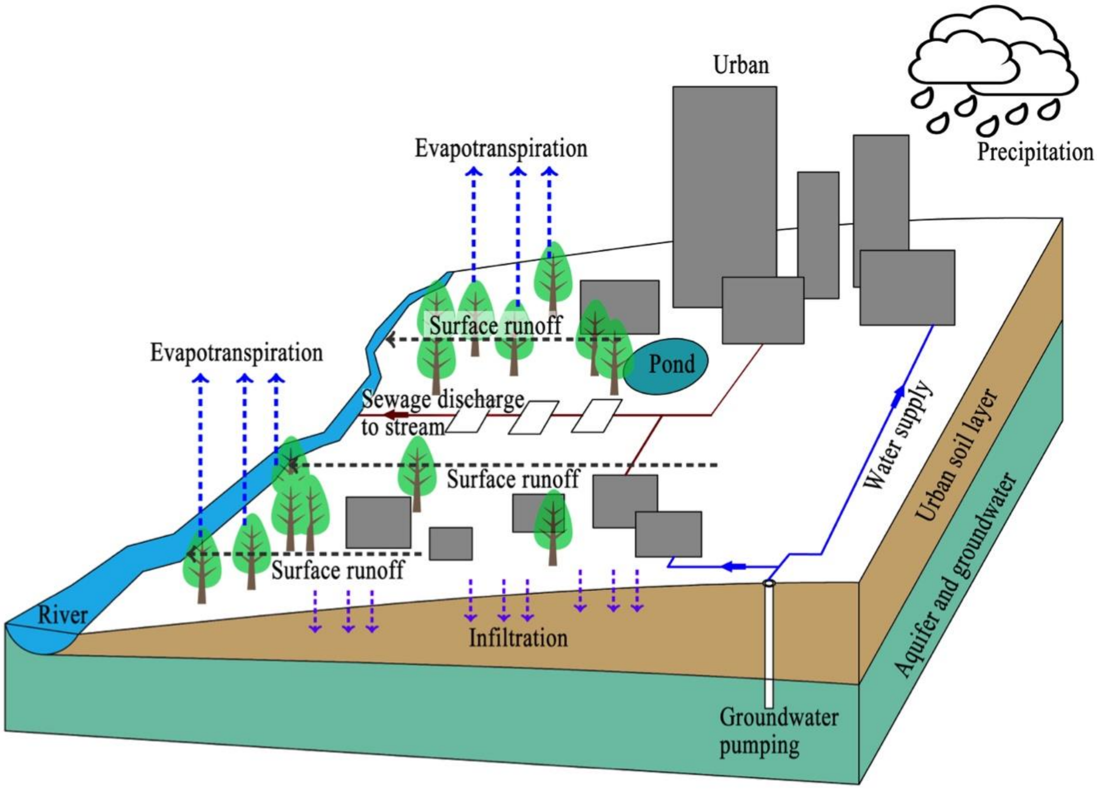


Fig. 2.1 The urban water cycle and different water forms in urban areas (schematic)

Profound damage in the ecological processes of urban stormwater is one of the most serious problems in the urban water cycle. Ideally, in a natural system, precipitation and runoff would soak into soils, recharging aquifers or returning to the atmosphere via evaporation (Woelfle-Erskine & Uncapher, 2012). However, urbanisation and the increasing extreme water excess and scarcity due to climate change adversely change the balance of the stormwater chain including precipitation, infiltration and evaporation in the urbanised catchments (Beecham et al., 2012; Dunnett & Clayden, 2007). The coverage of impervious surfaces (e.g. roads, pavements, roofs and parking lots, etc.) is dramatically increasing due to urbanisation.

For instance, in the United States, imperviousness was reported to range from over 50 % in multi-family communities, 70 % in industrial and commercial areas, to over 90 % in dense metropolises in urban areas compared to the low impervious coverage that may only be 1 or 2 % in rural areas (Schueler, 2000). Naturally, stormwater falls onto the earth is infiltrated depending on the role of vegetation and soils. However, these ‘sealed’ surfaces in urban areas are severely disruptive to the natural infiltration in that they do not let stormwater infiltrate into the soil, resulting in 75-80% less infiltration of surface waters to recharge the groundwater pools (Arnold & Gibbons, 1996). This can result in over three-fold larger volumes of surface runoff during a storm event compared to undeveloped landscapes (Burns et al., 2005; Michael & Keith, 2006). The increases in urban runoff causes flash floods (i.e. significantly increased runoff flows in a short time from the onset of a storm event), which results in more frequent flooding and heightened flood risks (DiBlasi et al., 2009; Pauleit & Duhme, 2000). Furthermore, on a global scale, urbanisation also results in increases in population and urban heat islands that increase temperatures in urban areas, which in return accelerate the water consumption the reduction in water availability (Golden, 2004).

Considerable seasonal and regional variation of changes in precipitation also leads to an increase of stormwater related disasters in urbanised watersheds (i.e. the total land area that drains to particular natural water bodies). For example, in northern Europe, climate modelling has predicted an increased incidence of severe precipitation

(Christensen & Christensen, 2003; Schröter et al., 2005), causing increased urban flooding (Arnell, 1999; Opdam & Wascher, 2004). Extreme events, such as the 2007 summer floods (Schumann et al., 2011) and the severe drought-flood fluctuation in 2012 summer in Britain (Ashley et al., 2013) are increasingly influencing the societal development and have thus drawn a fast growing public realisation of solutions for dealing with surface water (Ellis & Revitt, 2010). The sustainability of society and ecosystems is highly dependent on the security and supply of water resources (France, 2002). However, a general trend towards drier conditions with marked decreases in rainfall matching the warming pattern was reported in many regions in the world, such as in the south of Europe (Christensen & Christensen, 2003; Schröter et al., 2005). In the meantime, the increasing impervious surface coverage in urban development leads to a marked decrease in groundwater recharge and water tables, thus causing potentially adverse consequences for local communities and ecosystems. The lack of rainfall due to climate warming and the reduced infiltration in urban catchments results in depleted underground aquifers and very low river levels, thus severely influencing the availability of water supplies, which also makes the water conservation an extremely important topic in urban areas (Arnold Jr & Gibbons, 1996; Scibek & Allen, 2006; Vörösmarty et al., 2000).

The traditional stormwater solutions in cities are built in an industrial and engineering-based manner, which applies the construction of ever-larger underground drainage pipes that transport stormwater runoff as well as domestic and industrial wastewater to the nearest recipients (natural water bodies in most cases) as efficiently as possible (Allen, 2008; Chanan et al., 2010). There are two standard types of such systems:

1) Combined drainage system: mixed stormwater runoff and wastewater are drained in one pipe network and diverted to a wastewater treatment plant where various physical, biological or chemical processes are used to purify the mixed water, which is then discharged into the nearest water body (Fig. 2.2) (Burian et al., 1999).

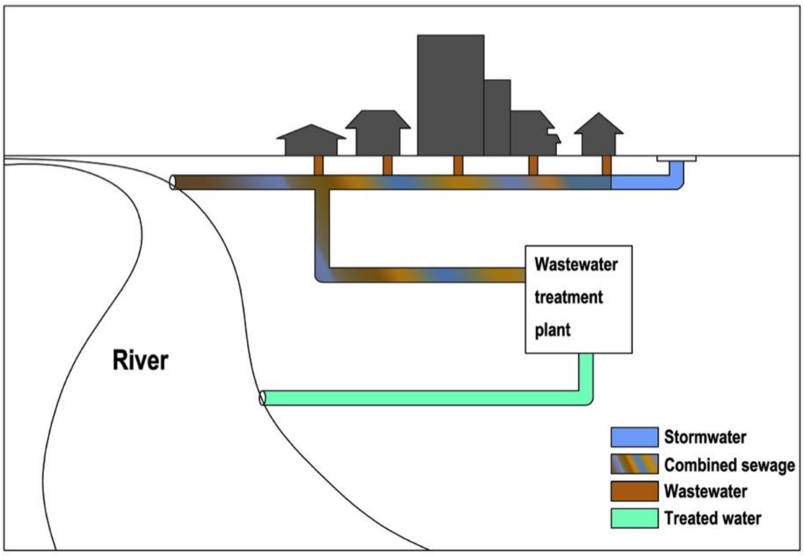


Fig. 2.2 The common combined drainage system in urban areas (schematic)

2) Separate drainage system: wastewater separation designed for water quality benefits that discharges treated/untreated stormwater to the receiving water bodies, whilst the wastewater is conveyed to the wastewater treatment plant (Fig. 2.3) (Black & Endreny, 2006). However,

only portions of urban drainage systems are able to separate the stormwater and wastewater due to high cost or physical limitations (Field, 1975; Tsay et al., 2003).

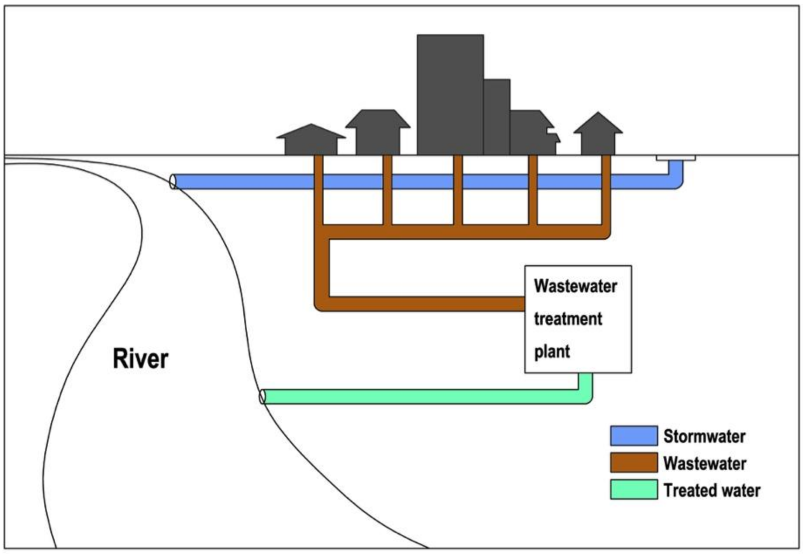


Fig. 2.3 The separate drainage system in urban areas (schematic)

Both types of the traditional urban drainage system drain stormwater rapidly to the nearest drainage point as efficiently as possible and have thus been beneficial for controlling flooding in cities for the past centuries (Berndtsson, 2010; Nelson, 2012). However, it is now understood that these conventional urban drainage systems are resource and energy consumptive for maintaining surface water quantity and quality (Kenway & Lant, 2012). Such systems are increasingly unable to cope with their aquatic burden during the storm events with increasing frequency and intensity (Allan, 2004; Eriksson et al., 2007; Wong, 2006a). This has raised concerns such as: (a) increasing disruptive flash-flooding in cities resulting from overflows in the receiving water bodies or when the piping systems are overloaded with stormwater runoff surges from impervious surfaces (Marsalek, 1998), (b) causing a notable reduction in infiltration, groundwater recharge and evaporation thus disrupt the urban water cycle (Butler & Parkinson, 1997) (c) discharge of untreated sewer or significant load of contaminants carried in surface runoff into recipients causing damage to local aquatic ecosystem (Taebi & Droste, 2004; Wong, 2006a) and severely impacting the quality of drinking water in cities (Lee et al, 2010). Such systems also fail to treat rainwater as a valuable resource for mitigating the water stress in the water-scare regions (Marsalek, 1998). The underground urban drainage infrastructures are invisible to city inhabitants and as a result there is limited public awareness of the importance of stormwater management, where people don't realise the piping controls of urban stormwater runoff is part of the problem (France, 2002). Additionally, the traditional urban water management facilities are often engineered or maintained with little or no consideration of the ecological, social or aesthetic qualities, which severely affect the public perception of the systems (Echols & Pennypacker, 2006).

There is, therefore, a constant requirement to increase the capacity of the urban drainage system and use stormwater as a valuable resource from the view of both drainage and water supplies. Alongside, from the urban design point of view, alternative urban stormwater solutions are expected to be visible and beautiful, which could increase the cities' amenity and quality of life, and provide a range of ecological and educational values (Echols, 2007).

2.4 Response to urban stormwater issues - sustainable stormwater management

2.4.1 Sustainable stormwater management implications

Major concerns related to the urban stormwater as noted previously have led to a considerably changed perception in civil engineering and landscape architecture in the past two decades, which turns away from the conventional developments, yet integrates ecosystem services and multifunctional land uses to restore the pre-development urban water cycle (Beck, 2013; Potter et al., 2011). The new approach – Sustainable Stormwater Management integrates Green Infrastructure (GI) for utilising the vegetation and soils' hydrology regulation services to retain, infiltrate, transpire and filter the urban runoff, so that the quantity and quality of urban stormwater runoff are controlled to mimic the predevelopment hydrology (Bortolini & Semenzato, 2009; LI, 2012). By using sustainable stormwater management, the necessary stormwater runoff reduction and purification are provided while adding other ecosystem services derived from the use of GI, vegetation and soils, such as water conservation, climate regulation, biodiversity conservation and aesthetic values to urban developments (Lloyd, 2001; Dunnett & Clayden, 2007).

Sustainable stormwater management has been developed from the critical points of the traditional urban drainage systems for their limited capacity and profound damages on the urban water cycle, and potentially offers numerous advantages over the traditional approaches (USEPA, 2012). Implementation of sustainable stormwater management requires availability of GI to enable the employment of the vegetation and soils. It works as landscape sponges to capture stormwater runoff from adjacent surfaces rather than piping the runoff as a nuisance. The retained runoff will either be infiltrated to recharge groundwater or returned back to atmosphere via evaporation, while the exceeded water will be slowly released to nearby public drainage inlets (Dussaillant et al, 2005; Endreny & Collins, 2009). Sustainable stormwater management is often applied at the source of stormwater generation, so that stormwater runoff can be effectively reduced on site to prevent the downstream urban catchments from the highlighted flooding risks (Carter & Jackson, 2007; Ferguson, 1990). Sustainable stormwater management facilities can increase groundwater recharge to ensure the sustainability of water supplies, while the evaporation of runoff is also enhanced to contribute to the balance of urban hydrology and provide evaporative cooling to mitigate the effects of urban heat island (Aravena & Dussaillant, 2009; Wong, 2006a).

Sustainable stormwater management is important to urban non-point source pollution control (Bedan & Clausen, 2009). Non-point source water pollution refers to the polluted runoff and wastewater that surge from diffuse urban drainage areas such as roads, roofs and parking lots (Trauth & Xanthopoulos, 1997). Sustainable stormwater management components are often designed to be very flexible for retrofitting within existing urban developments at any scale. The on-site installation of sustainable stormwater management can intercept and reduce the flow velocity to effectively trap particulate pollutants (suspended solids and trace metals) carried in the runoffs and sewage shed from urbanised catchments (Davis et al., 2009; Lucas & Greenway, 2011).

Sustainable stormwater management can provide an integrative approach linking the demands of urban drainage and urban design (Howe & Mitchell, 2012). With the removal of impervious caps from soil and the use of vegetation, the retrofit of sustainable stormwater management practices can minimise the lifeless urban hardscape. Their aesthetic attractiveness of otherwise grey urban environments is well recognised by professionals (Dunnett & Clayden, 2007). The increased amount of 'urban green' into the dense urban developments will then contribute to the wellbeing of urban dwellers (Dunnett & Clayden, 2007; Davis et al, 2009), which also allow biodiversity enhancement thus encouraging ecosystem services in urban areas (Kazemi et al., 2009; Kazemi et al., 2011). Sustainable stormwater management also makes the ecological processes of stormwater infiltration and filtration visible to city inhabitants. By making these ecological processes as key elements in urban landscape could provide the opportunity to improve the public awareness to the importance of stormwater management and the conservation of the urban water cycle (Ashley et al., 2011; USEPA, 2012). Significant economic benefits to communities and individuals can be derived from the cost-efficient installation and management of sustainable stormwater management compared to the traditional approaches (Andoh & Declerck, 1999; Coombes, 2002; Coombes et al., 2000; Kuhn & Frevert, 2010). Installation of sustainable stormwater management facilities could mitigate the urban heat island through the direct shading and indirect evaporative cooling provided by vegetation, and thus considerably reduce energy and fuel consumption for cooling (Dunnett & Clayden, 2007).

The adverse hydrologic effects resulting from climate change and urbanisation and the conventional water infrastructure resulted in some major rethinking of urban drainage systems in the late 1980s, where the first implementation of the sustainable stormwater management was initially introduced in public GI schemes in Prince George’s County, Maryland, USA (Davis et al, 2009; Prince George's County, Maryland, 1993). Its performance data to support the economic and environmental sustainability was delivered by the University of Maryland in 1997 (Dunnett & Clayden, 2007), since then the idea of sustainable water management gained increased ground. To date, the public acceptance towards the sustainable stormwater management has been growing significantly (Ashley et al., 2013).

The concept of ‘Sustainable Stormwater Management’ has been locally characterised in many regions in the world, and is referred to differently in different parts of the world (Ashley et al., 2013; Bortolini & Semenzato, 2010). For example, stormwater Best Management Practices (BMPs) in America (USEPA, 2000) and Low Impact Design (LID) in European countries (Dietz, 2007) were developed to use GI measures to sustain the storage, infiltration and evaporation of runoff that existed predevelopment. In the UK, the use of Sustainable Drainage Systems (SuDS) is encouraged by local authorities, which mainly focus on using the vegetated surface-based drains to discharge stormwater runoff while providing the services of infiltration and filtration (CIRIA, 2000; Mitchell, 2005). In Australia, the term 'Water Sensitive Urban Design' is commonly used, which integrates urban design with various technical solutions including GI and other engineering techniques to cope with the urban stormwater issues and any other environmental and social issues related to the urban water cycle (Howe & Mitchell, 2012; Khoo, 2009; Wong, 2006b).

The specific goals for the installation of the sustainable stormwater management would mainly include:

1) Restoration of the balance of the urban water cycle to approach the natural one and improve the efficiency and quality of stormwater management within urban developments,

2) Minimizing impervious urban surfaces to allow retention and infiltration of stormwater runoff close to source, and reducing runoff flow rate to reduce potential urban flooding risks,

3) Source control of non-point pollutions and protection of the urban water quality,

4) Recharging aquifers and rainfall harvesting for domestic use,

5) Reduction of untenable drainage facilities and related development costs,

6) Integration of stormwater management and the beautification and recreational amenity of urban areas,

7) Ecosystem services linked to air pollutant abatement, carbon sequestration, habitat augmentation and connectivity, biodiversity enhancement.

(Bortolini & Semenzato, 2009; Morison & Brown, 2011; Sieker & Klein, 1998; van Roon, 2007; Victorian Committee, 1999)

2.4.2 The contemporary components of sustainable stormwater management

Components for facilitating sustainable stormwater management have been actively developed in the past two decades. There are many types of sustainable stormwater management components depending on site-specific restrictions. Such components include rain gardens, bioswales, stormwater planters, green roofs, retention basins, filter basins, treatment wetlands, infiltration strips, permeable pavements and rainwater harvesting devices such as rain barrels, etc. (Liptan & Murase, 2000; Dunnett & Clayden, 2007). Most components expect the permeable pavements and rainwater harvesting devices to adopt the core idea of 'bio-infiltration' which refers to the use of the physical, chemical and biological properties of plants and soils to control the quantity and quality of stormwater runoff from urban areas within landscape measurements (Davis et al., 2009; Dunnett & Clayden, 2007). These components are site-specific and are developed as responses to particular stormwater management needs, as well as associated with advantages and disadvantages (Dunnett & Clayden, 2007). Appropriate selection of sustainable stormwater management components that appropriately adapt to the existing land use is vital to the efficiency of the system. Many of the components can serve multiple functions, such as the retention (i.e. reduction of stormwater volume that fall onto urban surfaces), infiltration (i.e. downward movement of rainwater through soil and bedrock), and evaporation of stormwater runoff and other environmental benefits (e.g. increasing wildlife value, reducing energy use and pollution, etc.) (Dunnett & Clayden, 2007). This introduction classifies the current sustainable stormwater management components into four categories according to their primary function: retention and infiltration, pollutant removal, rainwater reuse, and runoff conveyance.

Retention and Infiltration

Typically, the systems that are primarily used for stormwater retention and infiltration are often installed close to, or next to, buildings and impervious surfaces to temporarily store runoff diverted from roofs and adjacent pavements, and/or gradually infiltrate it into the ground (Dunnett & Clayden, 2007). In this way, a large portion of stormwater runoff is eliminated to lower the flooding risk in urban areas and the stress on public piping drainage systems. These components can be applicable in any scale, from private domestic gardens, through to larger-scale commercial landscapes (Dunnett & Clayden, 2007). These components are often installed in the public right-of-way, and are therefore perfect spots to show plantings with dramatic visual appeal among the other components (Steiner & Domm, 2012).

Rain Garden Rain gardens are shallow depressions employing designed vegetation with the appeal of gardens to intercept, retain and infiltrate runoff diverted from roof and adjacent surfaces (Fig. 2.4) (Woelfle-Erskine & Uncapher, 2012). Rain gardens are popular for domestic, commercial and municipal implementation to save the irrigation budget, while adding other ecological services such as pollutant sediment and absorption, biodiversity shifting, and aesthetic values (Dunnett & Clayden, 2007). Rain gardens are often combined with surface drainages or underdrain systems to discharge excess water rather than resulting in localised flooding (Steiner & Domm, 2012).

Fig. 2.4 Typical rain garden systems used to treat runoff diverted from a building roof and surrounding surfaces in London Wetland Centre, London, UK. Photo was taken by the author in August 2013.

Retention Basin Retention basins are vegetated surface storage basins that temporarily hold the stormwater runoff shed from the adjacent upland developments (Travis & Mays, 2008). The retained stormwater is either infiltrated within the vegetated basins or drained into nearby water bodies or additional conveyance systems to be converted to public drainage systems (Baek et al., 2014). Typically, there are two types of retention basins according to their normal ponding situation, i.e., dry and wet (Dickhaut et al., 2011). The dry systems often employ vegetation with higher density and soil amendments for greater porosity to infiltrate the detained water faster than the wet systems, and are typically dry between rainfall events (Fig. 2.5a) (Dickhaut et al., 2011). The wet systems are normally wet with plenty of retained water (Fig. 2.5b) (Dickhaut et al., 2011). The wet systems are often adopted to harvest rainwater for irrigation uses or to form water features in urban landscapes. Therefore, the wet systems would have either less vegetation coverage within their basins compared to the dry systems to have a slower soil infiltration to hold rainwater for a longer period, or to prevent infiltration with the use of impervious liners. Both systems can be incorporated into urban developments for trapping the pollutant loads carried in runoffs, as well as gaining aesthetic amenity and biodiversity benefits (Bhaduri et al., 1995; Dickhaut et al., 2011).

(a) **(b)**

Fig. 2.5. Retention basins

a: Retention basin system that stays dry between rainfall events, Berlin, Germany. Photo was taken by the author in April 2012.

b: A typical wet retention basin used to hold runoff diverted from highway, Orlando, Florida, USA. Photo was taken by the author in May 2013.

Green Roof Green roof is the vegetated, multi-layered structure installed on the rooftop of a building (Nagase & Dunnett, 2012). A green roof uses its vegetation, growing media and sometimes an additional water storage layer (e.g. PVC modules with a number of small cups to retain water) to reduce the amount of runoff generation from storm events (Fig. 2.6) (Dunnett & Kingsbury, 2004; Dunnett & Clayden, 2007). Big size solids are filtered with a filter mat underlying the vegetation and soils, while the excess stormwater is discharged through the drainage layer (e.g. normally comprised with pebbles) (Fig. 2.6) (Dunnett & Kingsbury, 2004; Dunnett & Clayden, 2007). The existing rooftop is prevented from being saturated by overlying waterproof materials (Fig. 2.6) (Dunnett & Clayden, 2007). Green roofs are either extensive featured with light, thinner substrates and planted with succulents or perennials (Fig. 2.7a), or intensive planted with deep-rooted vegetation (e.g. trees and shrubs) on heavy and thicker growing medium (Fig. 2.7b). Green roofs could contribute greatly to the amenity and aesthetic value of buildings, and frequently employ native plant species promoting urban biodiversity (Dunnett et al., 2011).

Fig. 2.6. The typical multi-layered green roof system (from top to bottom: vegetation, growing medium, filter mat, water storage and drainage layer, root barrier and waterproof layer). Photo was taken by the author in June 2013.

(a) **(b)**

Fig. 2.7. Extensive green roof featured succulents and perennials on light growing medium (a), and intensive green roof featured woody plants with heavy growing medium (b), Sheffield, UK. Photo was taken by the author in March 2012 and March 2015, respectively.

Stormwater Planter Stormwater planters are contained vegetation areas fitting with building façades to catch and infiltrate the rooftop runoff via downspouts from rooftops (Fig. 2.8) (Liptan & Murase, 2000). This system is very adaptive and could be retrofitted to where infiltration is desirable in the dense urban developments while adding aesthetic appeal to the adjacent landscape. The retention area of a stormwater planter is typically shallow and small, thus stormwater runoff can only be temporarily detained in the system for a rather short time (Liptan & Murase, 2000). Runoff is either absorbed and infiltrated through the vegetation and soils in the stormwater planter, or conveyed to the approved runoff disposal inlets using embedded discharge pipes (Dunnett & Clayden, 2007).

Fig. 2.8. Stormwater planters that are well-fitted with building façade, Windermere, UK. Photo was taken by the author in March 2013.

Permeable Paving Permeable pavements include pervious concrete, asphalt or resin bound pavers that allow the downward movement of stormwater runoff through their surfaces into the ground (Fig. 2.9) (Dickhaut et al., 2011). Permeable paving directly promotes the runoff infiltration on-site thus controlling the runoff generation at its source (i.e. vehicular or pedestrian traffic spaces), as well as filters suspended solids from surface water without altering the existing land use (Dickhaut et al., 2011).

Nevertheless, the soil bed below permeable paving is often compacted, so that its water table could quickly reach a high level to prevent the precipitation from being absorbed into the ground (BASMAA, 1999). Therefore, permeable paving needs underlying grade gravel beds to detain the water until the soil field capacity gains sufficient restoration to absorb water (BASMAA, 1999). It also requires fairly intensive maintenance because suspended solids would block the open pores thus also affecting its longevity (BASMAA, 1999).

Fig. 2.9. Permeable paving comprised by resin bound aggregates (SureSet UK Ltd., Warminster, UK) which allows great proportions of water to pass through. Windermere, UK. Photo was taken by the author in March 2014.

Runoff conveyance

Runoff is transported in traditional stormwater solutions in impervious drainage pipes, thus resulting in little infiltration and evaporation. Conveyance components of sustainable stormwater management make the runoff movements visible and employ vegetation and soils to encourage infiltration and evapotranspiration, until the remaining part of the runoff reaches the approved runoff disposal points.

Bioswales Bioswales are linear vegetated channels to transport and temporarily store stormwater runoff diverted from adjacent surfaces (Fig. 2.10) (Dunnett & Clayden, 2007). Bioswales use permeable bases and vegetation for runoff infiltration during transporting stormwater runoff to approved public sewers or other associated sustainable stormwater management components. It can be flexibly adapted to any urbanised scenarios and provides hydrologic links among landscape features, urban developments and drainage systems, and add aesthetic values to neighbourhoods (Steiner & Domm, 2012).

Fig. 2.10. Bio-swale conveying and infiltrating runoff from an upland allotment to downstream watershed, Sheffield, UK. Photo was taken by the author in October 2013)

Infiltration Strip Infiltration strips are sloping vegetated areas that intercept runoff from adjacent surfaces or disconnected downpipes, and shed it to other conveyance systems, nearest water bodies or treatments (Fig. 2.11) (Blanco-Canqui et al., 2004). Wider infiltration strip is expecting to deliver better hydraulic performance as it spreads the runoff over a larger surface thus breaking the flow to allow sufficient runoff infiltration, pollution absorption and decrease of runoff flow rate before runoff reaches the downstream watershed (Dunnett & Clayden, 2007).

Fig. 2.11. Infiltration strip that shed car park runoff to permeable ground, Sheffield, UK. Photo was taken by the author in April 2014.

Runoff pollution filter

Runoff purification is necessary before stormwater is infiltrated to recharge groundwater, discharged into natural water bodies or used in domestic water services to prevent disruption of the aquatic security and human health by the non-point pollution associated with surface runoff. Some of the sustainable stormwater management components apply pollutant removal as their primary goals and have advanced abilities of pollutant and toxic substances removal among the other components.

Filter Basin Filter basins are shallow vegetated depressions, which typically employ engineered soils and specially enhanced vegetation to remove pollution and downstream runoff (Fig. 2.12) (Lazareva & Pichler, 2010). Filter basins employing designed plant communities with metal-accumulating plants to hold the toxic accumulation in the nutrient-rich or polluted urban stormwater runoff, and this vegetation is often very dense to maximise the pollutant removal benefits (Salt et al., 1995; Lazareva & Pichler, 2010). Soil microorganism in filter basins also help to break down the toxic chemicals carried in runoff shed from urbanised catchments (Dunnett & Clayden, 2007).

Fig. 2.12. Typical filter basin with dense planting, Olentangy River Wetland Research Park, Columbus, USA. Photo was taken by the author in September 2012.

Treatment Wetlands Treatment wetlands are constructed wetlands that employ deliberate assembled plant communities that are capable of reducing nutrient input associated with polluted urban runoff to ensure the downstream aquatic security (Mander & Mitsch, 2009). It is often adapted into developments near natural water bodies to provide solid pollutant removal benefits and mitigate water scarcity in wet seasons, as well as adding aesthetic amenity while increasing urban biodiversity (Fig. 2.13) (Green & Upton, 1994; Brix & Schierup, 1990; Shutes, 2001).

Fig. 2.13. Treatment wetland featured dense reed beds and species-rich upland perennial planting and woodland in Rotherham Centenary Park, Rotherham, UK. Photo was taken by the author in August 2013.

Stormwater harvest

Rainwater harvesting evolving from arid regions has been widely adopted into contemporary developments and landscapes (Dunnett & Clayden, 2007). It uses devices such as the rain barrels, large containers or embedded cisterns to capture runoff from roofs and other surfaces (Fig. 2.14), so that the total runoff is reduced through storage (Dunnett & Clayden, 2007). This stored part of runoff could be used for non-portable applications (i.e. garden irrigation, toilet flushing and washing machine, etc.) in dry seasons (Dunnett & Clayden, 2007).

Fig. 2.14. Runoff harvesting barrels fitted with roof gutter in Manor community, Sheffield, UK. Photo was taken by the author in October 2013.

2.4.3 Opportunities and application development of the sustainable stormwater management

There is a timely opportunity for introducing sustainable stormwater management as the alternative option to the traditional approaches for the minimisation of serious flooding risks following heavy storm events and providing the various ecological benefits (Trowsdale & Simcock, 2011). Sustainable stormwater management systems should be appropriate to the context and therefore range according to the spatial scale of urban design. For example, in the local community context, both stormwater management and living environment could be improved by retrofitting of sustainable stormwater management components (Fig. 2.15). The introduction of green roofs could capture rainfall that falls onto building rooftop and release the excess water into rain harvesting devices (e.g. rain barrels) for domestic uses to gain cost efficiency.

Preferred vegetation could be designed with suitable soil mixtures to form domestic/municipal rain gardens and bioswales to govern the ecological treatment of stormwater runoff surges from hard surfaces and provide visual amenity. Pedestrian priority areas could adopt permeable pavements to support drainage and allow more infiltration to increase the recharge of urban aquifer. The adaptation of sustainable stormwater management systems could support the growth and resilience to climate change and flooding risks in the long-term for larger scale (e.g. commercial scale) urban developments. At the larger scale, the goal of sustainable stormwater management design is to reduce the overall imperviousness associated with the proposed development, to optimise the public amenity and to maximise the value of land by allowing various activities. This could be done by the introduction of the various sustainable stormwater management components (Fig. 2.16). For example, commercial buildings can provide the opportunities to establish large areas of green roofs and stormwater planters to capture and filter the rooftop runoff. The large impervious area of park lots could be replaced with permeable surfaces for flood mitigation. Rain gardens and bioswales could be retrofitted to improve the public aesthetics with the appeal of dynamic vegetation, but could also capture and purify the runoff from roads and parking to mitigate the pressure of public drainage systems during storm events and protect the downstream aquatic ecosystem. The use of retention basins could harvest hard surface runoff for providing water playing features in public spaces.

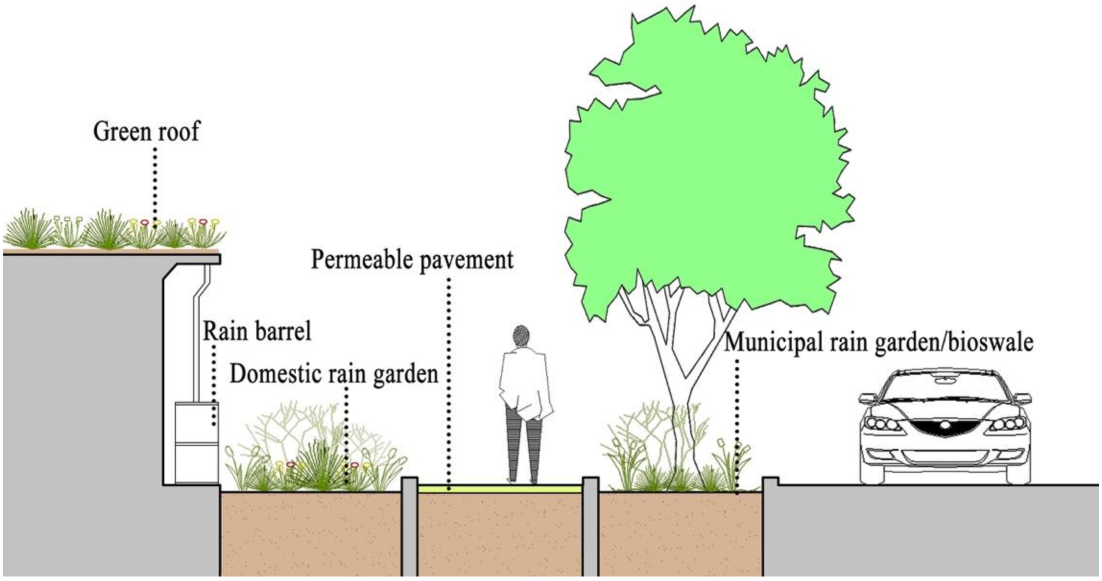


Fig. 2.15. Conceptual sustainable stormwater management system for neighbourhood design

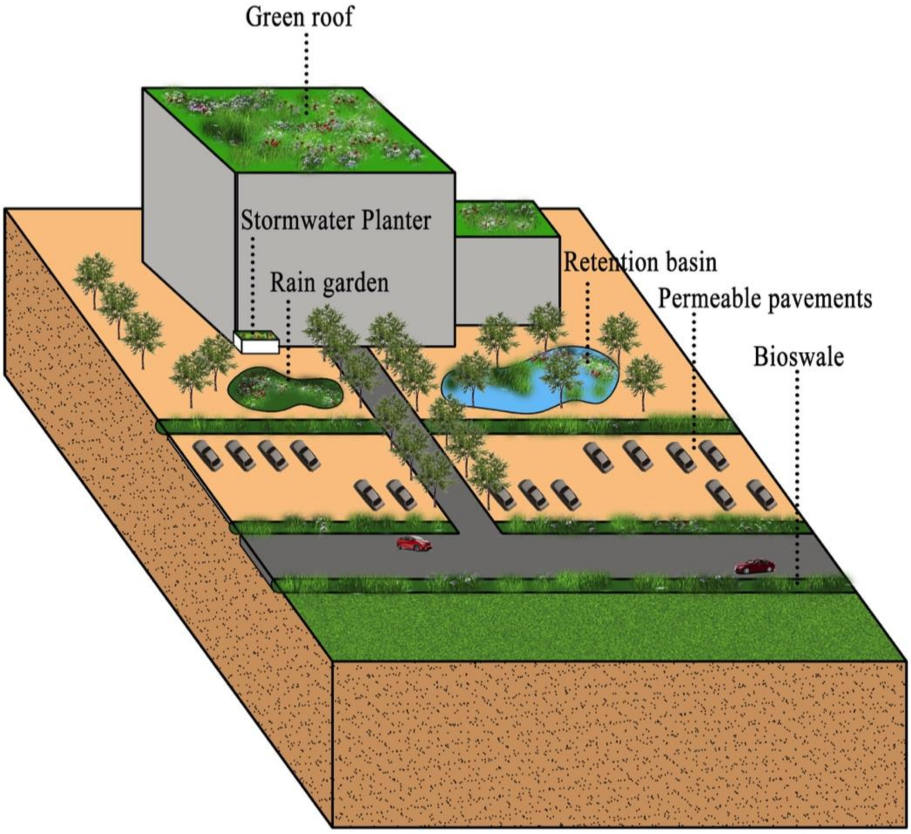


Fig. 2.16. Conceptual sustainable stormwater management system for commercial development

Sustainable stormwater management has gained popularity in application in Australia and the USA in particular and is progressively emerging in the UK and other countries such as Germany, New Zealand and Singapore (Ashley et al., 2013; Brown and Hunt, 2011; Wong, 2010). Brown and Clarke (2007) and Brown et al. (2008) suggested that the sustainable stormwater management implementation in most regions of the world is in a learning-by-doing period of trials of the new ideas, whereby pilot projects are stabilised to attract the socio-political capital and public awareness through demonstrable results. Introduction of the development of sustainable stormwater management locally in the UK, and in the USA and Australia is presented below.

United Kingdom

The United Kingdom has a very positive stance for taking the sustainable stormwater management vision of urban development (CIRIA, 2013). Sustainable stormwater management is emerging in the UK (Burton, 2012; Shaffer et al., 2012) and there is a growing public awareness and preference on it compared to the traditional pipe drainage systems in new urban developments (Duffy et al., 2012; Elmer & Fraker, 2012). A recent questionnaire investigation showed a good acknowledgement of sustainable stormwater management in that 68% respondents in total 207 professionals involved in urban planning and urban design in the UK were familiar with it (CIRIA, 2013). In Europe, particularly in the UK, legislation for sustainable stormwater management is advanced in the world. A variety of regional or local legislations for all issues concerning urban stormwater were developed in the

UK under the guide of the *Water Framework Directive of the European Union* (European Parliament and of the Council, 2000), which put a high priority on the protection and revitalisation of the entire urban water cycle in urban areas. Sustainable stormwater management in the UK has been included in the development policies for more than half of the local authorities (Woods-Ballard, 2012). There are many British principal instruments that require sustainable stormwater management to be specifically considered at every level of flood management and the mitigation of negative environmental impacts resulting from urbanisation, which include the *Code for Sustainable Homes* (DCLG, 2011), *Technical Guidance to the National Planning Policy Framework* (DCLG, 2012), *Water Cycle Studies Guidance* (EA, 2009), *Surface Water Management Plan Technical Guidance* (Defra, 2012), *Planning Policy Statement 25 for Development and Flood Risk* (CLG, 2006) and the *Development and Flood Risk: A practice guide companion to PPS25 'Living Draft'* (CLG, 2007).

There are a few pilot examples of the adoption of sustainable stormwater management in the UK. For instance, a series of rain gardens within 500 metres of the River Lee were installed in the Kingsmead Estate in London by local residents and the charity Groundwork London (2012) to prevent the river from the risk of flooding and being adversely affected by urban pollution. There are a few pilot examples involving the integration of different sustainable stormwater management components to maximise the ecosystem services that were introduced in the UK for exhibition and public education. For example, two demonstration rain gardens in NEC Gardeners'

World 2000, Birmingham (Dunnett & Clayden, 2007) and the London Wetland Centre applied the similar idea that using rain barrel and stormwater planters to harvest excess water from roof gardens for domestic use or irrigating the terrestrial rain gardens planted with species-rich plantings, while timber rills were used to circulate rainwater within the yards so that the ecological water cycle could be experienced visually and even be interacted. Green roof installations such as the roof gardens on the rooftop of Moorgate Croftes, Rotherham (Dunnett & Clayden, 2007) and the Sharrow Primary School, Sheffield, demonstrated the very advanced vegetation technology in the UK. These cost-efficient plantings integrated the services of runoff infiltration biodiversity conservation, while adding aesthetic and recreational values (Nagase & Dunnet, 2012).

However, in the UK, the adoption of sustainable stormwater management is still underutilised at local level and is in an early stage of legal promotion and field experiments, where professionals and individuals may underderestimate its functionality and feasibility (Ashley et al., 2013). It is partly because people are unfamiliar with the technologies involved in sustainable stormwater management, and partly because there are rather limited demonstrable projects for attracting mainstream institutional legitimacy (Ashley et al., 2013; CIRIA, 2013). Ashley et al. (2013) also claimed several other barriers to adopt sustainable stormwater management for the planning authorities, investors and dwellers who are accustomed to the conventional stormwater solutions, which include: (a) the lengthy and complicated approval shifting and licensing process for making new strategies for sustainable stormwater management, (b) the additional costs to implement sustainable stormwater management, (c) organisational resistance due to the risk aversion and limited regulatory incentives, and (d) the uncertain engagement of critical stakeholders.

United States

In the USA, the long term development towards the sustainable stormwater management started since 1987 when the Clean Water Act (CWA) was amended to require the US Environmental Protection Agency (USEPA) to establish the National Pollutant Discharge Elimination System (NPDES) to develop sustainable stormwater management components to control the quantity of surface runoff in urban areas and to address the non-point sources of runoff pollution (Adler et al., 1993; Andreen, 2004; USEPA, 2002; USEPA, 2008). To date, the adoption of Best Management Practices (BMP) or the Low Impact Development (LID) in the USA have been successful in terms of the water scarcity and excess mitigation (Coffmann, 2000; Zhen et al., 2004; Dietz, 2007; Dietz & Clausen, 2006).

Compared to the limited implementation in the UK, sustainable stormwater management developments in the USA are implemented more on a local level. Some of the American local authorities encourage city-wide programs to utilise cities to work as coherent sustainable stormwater management systems. For instance, in Philadelphia, a large number of sustainable stormwater management components including green roofs, rain gardens, permeable pavements and bio-swales are retrofitted in existing urban developments to infiltrate the first 1-inch rainfall from all directly connected impervious surfaces (PWD, 2006) and reduce combined sewer overflows across their watershed (PWD, 2012). Portland and Chicago are the two cities that first embraced the sustainable stormwater management where many pilot projects can be found in public sites (Dickhaut et al., 2011). For example, Chicago adopted the Green Alley Program to enhance the urban infiltration through the use of permeable pavement and retention basins, which prevented over 1900 miles of alleys which were without connections to the combined or storm sewer system from frequent flash flooding (CDOT, 2007). The From Grey to Green project in Portland developed inherent sustainable stormwater management that retrofitted green roofs, rain gardens, stormwater planters, bio-swales and constructed wetlands as a holistic system to counteract the periodic urban flooding during heavy rainfall events while greatly enhancing communities' greenery (BES, 2008).

Implementation of sustainable stormwater management in the USA is particularly incentive-based so that good surface runoff management is now considered 'business as usual' (Dickhaut et al., 2011). For instance, the Tanner Springs Park, Portland, uses retention basins that mimic natural wetland habitats in a commercial-residential area, which integrates recreational uses and sustains the property value (Dickhaut et al., 2011). The incentive-based sustainable stormwater management implementation successfully gained enthusiastic public involvement, for instance, in Somerset, New Jersey, USA, the local communities proudly installed a series of rain gardens and rain barrels for the collection and reuse of rainwater, as well as keeping polluted water from the impervious urban watershed out of the local streams under the lead of the New Jersey Water Supply Authority (NJWSA).

Most sustainable stormwater management activities in the USA are happening at the local level, whereas the federal authorities such as the USEPA are not actively pursuing regulatory control thus effective guardianship is limited at the country and city level (Roy et al. 2008). Compared with the increasing willingness and the advanced legislations for taking up the sustainable stormwater management vision in the UK federal authorities, properly designed policies are needed in many US regions to help mitigate the increased costs caused by transferring maintenance costs to different parties and the additional costs for professional training and education (Roy et al., 2008).

Australia

Sustainable stormwater management was developed in Australia because of the major concerns of the deteriorating health of watercourses (Brown & Clarke, 2007; Wong & Brown, 2009). The Urban Stormwater Best Practice Environmental Management Guidelines (Victorian Committee, 1999) led to a localised framework of sustainable stormwater management for urban planners in Melbourne, which was fairly agreed as a starting point for many of the recent sustainable stormwater management projects in Australia.

Similar to the US situation, in Australia, the rather limited regulatory incentives on a national level and the financial shortage are two main barriers for the implementation of sustainable stormwater management in Australia (Farrelly & Brown, 2011). Conversely, a variety of local characteristised sustainable stormwater management strategies are strongly recommended for new urban developments in different states and cities, while related programmes are supported by active practitioner communities (COAG, 2004; Wong & Brown, 2009). In Melbourne, sustainable stormwater management systems have been thoroughly adopted into urban areas for stormwater harvesting, treatment and reuse (Dickhaut et al., 2011). For instance, sustainable stormwater management components are encouraged such as implementing green roofs and retention basins for temporarily detaining runoff and providing runoff pollutant removal, whereby the treated stormwater is directed to the rainwater harvesting tanks for reuse such as toilet flushing and landscape watering (Dickhaut et al., 2011; Wong, 2006b). The integration of various sustainable stormwater management components such as the constructed wetlands, bio-swales, and rain gardens are widely installed within landscape elements associated with public buildings at a local and regional scale (Wong, 2006b). Other Australian cities including Sydney, Brisbane and Perth have also maintained active movements in the implementation of sustainable stormwater management (Taylor, 2010).

2.5 Overview of rain garden design aspects

Rain gardens are small-scale sustainable stormwater management components that are land-based and consist of excavated shallow depression structures that are backfilled with a combination of vegetation and soils (Dunnett & Clayden, 2007). The use of such features in residential, commercial and municipal developments is increasing rapidly because of their numerous ecosystem services compared to the conventional stormwater approaches (Davis, 2005; Trowsdale & Simcock, 2011), and their cost-effectiveness, aesthetic values, as well as the flexibility in terms of size and location among other sustainable stormwater management components (Steiner & Domm, 2012; Wong & Brown, 2009). Rain garden is therefore the main focus of this PhD study. Below is a review of the important design aspects of typical rain gardens including their implications, diversity, configuration and sizing criteria.

2.5.1 Rain garden implications and diversity

As noted previously, the two key elements of a rain garden are the vegetation and soil to retain the rainwater conveyed from adjacent impervious surfaces and allow natural infiltration, which is strengthened by vegetation and potential soil amendments. Rain gardens may not be very visually different from the vegetated beds in ordinary gardens at first glance, however, most rain gardens are built in shallow depressions that are backfilled with soils or other permeable substrates (Fig. 2.17) (Steiner & Domm, 2012). The shallow depression of a rain garden could harvest rainwater for a short duration to contribute to the reduction of surface runoff in urban development and provide vegetation with sufficient irrigation. There would be a lag time between the onset of runoff and the rainfall event in rain gardens. The lag time occurs because it takes time not only for the excess water to reach beyond the ponding limitation of the depressions of rain gardens, but also for the rainwater to flow through the soil depth (Steiner & Domm, 2012). This lag time may allow more time for rescue and evacuation in a severe flooding event (Steiner & Domm, 2012). Therefore, the most important advantage of the application of rain gardens over ordinary gardens is the combination of stormwater runoff reduction to prevent urban developments from flash flooding and filtration (e.g. trapping and removal of the pollutants carried by urban runoff) with the appeals of a garden (Dunnett & Clayden, 2007). It is noticeable that practical rain gardens are required to either have relatively porous soil or to be connected to an urban drainage system to completely dewater within 24 to 96 hours (Davis et al., 2009; Steiner & Domm, 2012). Therefore, rain gardens are not ponds or wetlands, which only hold standing water for a short period.

Fig. 2.17. Typical rain garden with vegetated shallow depression in residential area, USA. Picture source: http://water.epa.gov/infrastructure/greeninfrastructure/images/gi_raingarden.jpg.

Design of rain gardens can vary in form and function according to context. Unlike ordinary gardens that are placed within reach of water hose to enable regular irrigation, rain gardens are irrigated by natural rainfall. Therefore, rain gardens should be built close to the source of stormwater runoff. Domestic rain gardens are often connected with building downspouts to treat the runoff diverted from a rooftop, or be placed in a low area so that stormwater runoff surging from adjacent surfaces is conveyed to the excavated depressed basins of the rain gardens by the effect of gravity (Fig. 2.17). It is worth noting that the domestic rain garden should be at least 3 metres away from the building’s foundation to prevent water leaks in the basement or rot in wooden floors (Steiner & Doom, 2012). Rain gardens are also commonly seen as lower-lying basins fitted beside paved public spaces such as roads, concourses, and parking lots, etc. to provide on-site runoff treatment (Fig. 2.18). The rain gardens located beside roads and parking lots often have impermeable berms rising upon the road surfaces and greater ponding depths to hold excess runoff shed from the impervious surfaces until it is absorbed into soils and substrates (Fig. 18). Such rain gardens have curb cuts to accept runoff inflow from roads or parking lots (Fig. 2.18). To prevent flash flooding and potential long periods of waterlogging in urban developments, rain gardens may be melded into existing urban drainage systems (Steiner & Doom, 2012). Embedded perforated underdrain pipes or overflow drain inlets are the two most widely used measures to connect rain gardens and public drainage systems for a faster draining of the excess stormwater runoff in urban areas (Fig. 2.19).

Fig. 2.18. Rain garden used to infiltrate highway runoff, Columbus, USA. Photo was taken by the author in October 2012.

Fig. 2.19. Public drain inlet placed on the marginal area of rain garden to quickly drain the excess surface runoff, London 2012 Olympic Park, London, UK. Photo was taken by the author in August 2013.

2.5.2 Configurations of a typical rain garden

Installation of rain gardens can be cost-efficient and simple, but they need to employ appropriate layers to maximise their benefits in stormwater treatments. A typical rain garden has a few layers as shown in Fig. 2.20. The storage capacity of a rain garden is mainly determined by the depression of the rain garden which is governed by its surface area and the ponding depth. The allowable ponding depth is defined as the vertical distance from the lowest excavated depression level to the ground level. The desired ponding depth must be at least 5 cm (Woelfle-Erskine & Uncapher, 2012). However, it may vary with specific design proposals and site conditions such as the depth of a design storm to be captured in the rain garden, the infiltration rate of soil or substrate and the slope of the adjacent drainage area, etc. For example, a deeper depression may be required to be able to store more runoff for a large storm event, or be required in a site with relatively poorly infiltrating soils.

Mulch can be applied over the top of rain garden soils for weed control and provide nutrients for vegetation (Dunnett & Clayden, 2007). The pre-treatment filter strip that consists of vegetation and soils is the most important zone to capture and filter stormwater runoff (Woelfle-Erskine & Uncapher, 2012). The depth of the rain garden soils or substrates varies with the specific intention of the design. For example, increasing the depth of soil could increase the capacity of the rain garden (Steiner & Domm, 2012), as there would be more soil available to hold water. Soil infiltration is inversely related to the ratio of fine particles (e.g. silt sized between 0.05 mm and 0.002 mm and clay sized less than 0.002 mm) to the media, where soil with a smaller ratio would typically have a larger infiltration rate (Hinman, 2009). Poorly drained soil (e.g. heavy clay or other poorly draining materials) would cause runoff to pond on the surface rather than be soaked into the ground. In practices, native soil that drains less than 0.25 cm/hour is recommended to be replaced by well-draining soil or mixed with other porous substrates to allow increase in infiltration in order to avoid long-term waterlogging (PADEP, 2006; Woelfle-Erskine & Uncapher, 2012). A variety of suggestions of rain garden soil mix were given in current research and design manuals. For example, Sickles et al. (2007) suggested a mix consisting of 61% sand, 16% silt and 23% clay, while PADEP (2006) suggests any soil media mixed with sand and topsoil that has a clay content under 10% would be ideal to allow water to flow through quickly. However, it is also worth noting that soil with higher silt and clay contents would potentially hold more water and thus benefits plant growth during dry spells (Barrett et al., 2013).

A clean sand layer and a pea gravel (particles with a diameter of 2 to 4 mm) layer of 7.5 cm minimal depth could be embedded under subsoil for providing further filtration of runoff (Woelfle-Erskine & Uncapher, 2012). The layer laid beneath the sand and pea gravel layers is the drainage layer of 15 cm minimal depth that comprises washed rocks or gravel sized between 2 to 4 cm (Woelfle-Erskine & Uncapher, 2012). Commercial or municipal rain gardens would normally have underdrain pipes or overflow pipes to prevent the practices and properties from flooding during a large storm event (Steiner & Domm, 2012). The underdrain pipes are perforated and are covered with the gravel drainage layer (Woelfle-Erskine & Uncapher, 2012). The inlet of the overflow pipe would be adjusted to a certain height to quickly dewater the potential overflow in the rain garden and may thus alter the ponding depth. The underdrain pipes and overflow pipes should be connected to the public drainage system to discharge the excess water in rain gardens (Steiner & Domm, 2012).

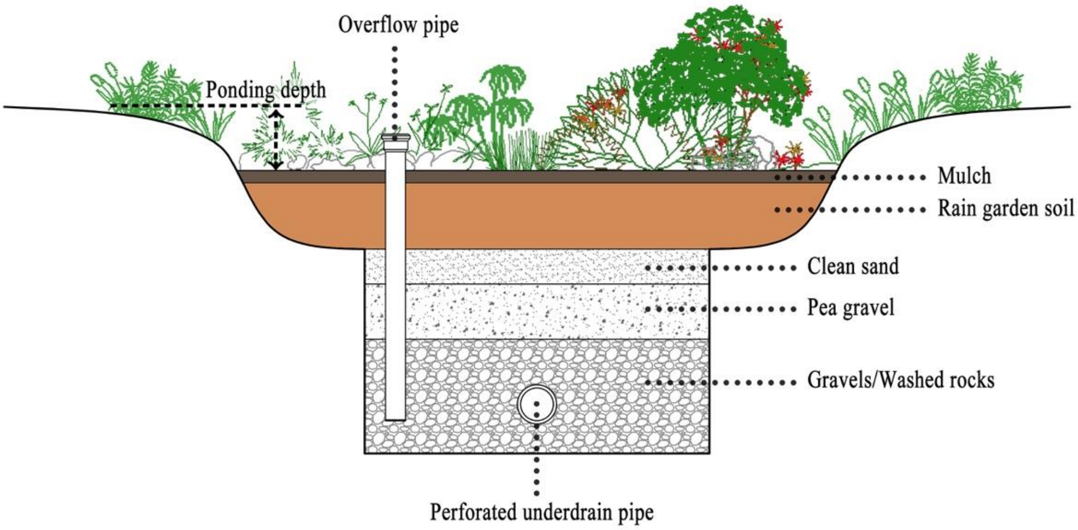


Fig. 2.20. Typical layers in the structure of a rain garden (schematic)

2.5.3 Sizing criteria

As noted previously, rain gardens are flexible in size and shape. Due to the insufficient availability of spaces in urban developments, rain gardens are often retrofitted into existing landuse with being properly sized, which is important to the success of design especially for rain gardens (Campbell et al., 2013). However, many rain gardens are simply fitted into remaining space available on site rather than being properly sized for stormwater benefits (Campbell et al., 2013). There is little academic literature discussing the sizing methods of rain gardens. In practice, the original minimum sizing criteria for a rain garden was based on theoretical considerations of deign parameters including the width, length, and ponding depth (Clar & Green, 1993). A wide range of methodologies have been developed for sizing rain gardens (Davis et al., 2009), while these methods are variable that often confuses people. Critical discussions are provided below for two of the most widely adapted sizing methods in current rain garden design.

Most North American guides recommend rain gardens to have the capacity to capture runoff in a storm event with up to a 1-inch (25.4 mm) rainfall (Steiner & Domm, 2012). In most instances, setting the capacity of the system for an inch of rainfall is reasonable, for instance, 1-inch rainfall is greater than the estimated 1 in 10 year event (21.94 mm) locally in Sheffield, UK (NERC, 1999), which can be considered as a significant event (e.g. event with a return period that is greater than one year, Stovin et al., 2012). Examples such as 'Rain Gardens: A how-to manual for homeowners' (Bannerman & Considine, 2003), 'Rain Gardens Across Maryland' (Worcester County Department of Comprehensive Planning, 2008), 'Rain Gardens: A Manual for Central Florida Residents' (D'Abreau, 2010) and 'Rain Garden Manual for Homeowners: Protecting Our Water, One Yard at a Time' (Northeast Ohio Public Involvement Public Education Committee, 2006) have developed their way of sizing rain gardens for completely holding and infiltrating the 1-inch rainfall, in which the calculation of the rain garden size is based on three key factors including: (a) total drainage area (i.e. the area of surfaces that generates runoff and is drained to the rain garden), (b) ponding depth, and (c) the texture of soils in the rain garden.

In this method, once the location of the rain garden is determined, the total drainage area needs to be calculated in order to help find out the approximate size of the rain garden. The total drainage area equals the sum of roof drainage area and contributing ground drainage area (i.e. the area of ground surfaces that generates runoff and is diverted to the rain garden) (1). It suggests that the proportion of the downspouts that directly feed water to the rain garden multiplied by the total rooftop area of the building equals the roof drainage area (2).

$$\text{Total drainage area} = \text{Roof drainage area} + \text{Contributing drainage area} \quad (1)$$

$$\text{Roof drainage area} = \frac{\text{Number of downspouts linking to rain garden}}{\text{Total number of downspouts}} \times \text{Total roof area} \quad (2)$$

Next, estimate the ponding depth of the rain garden's storage area. The ponding depth is strictly relevant to the slope of the ground where the rain garden is sited. More runoff may be converted into rain garden depression when the adjacent surface has a steeper slope and thus may require a greater ponding depth. Table 2.1 gives an idea of how deep the depression should be relating to the slope.

Table 2.1. Suggested ponding depth in rain garden (adapted from Worcester County Department of Comprehensive Planning, 2008)

Slope	Ponding depth (cm)
< 5%	10-15
5-7%	15-18
8-12%	20

The third step is to identify the Rain Garden Size Factor. The Rain Garden Size Factor is a rapid rain garden size calculation tool suggested by engineers. The Rain Garden Size Factor is determined from the distance between downspouts and rain garden, ponding depth and soil type (Table 2.2). Although none of these technical manuals have provided reliable references on how the Rain Garden Size Factors were obtained, these parameters are assumed to be relatively reliable as they demanded the three facts that: (a) less amount of runoff would be collected with longer distance between rain garden and downspouts, (b) depression with greater depth would hold more runoff than depression with same length and width but less depth, (c) rain gardens in a poorly drained area must have a larger depression than the rain garden built in well-drained soil to hold excess water. There are three main types of soils suggested in these manuals including sandy soil (gritty and coarse soils), silty soil (smooth but not sticky soils) and clay (sticky and clumpy soils). The result of the rain garden size will be the total drainage area multiplied by the Rain Garden Size Factor (3).

$$\text{Rain garden size} = \text{Total drainage area} \times \text{Rain Garden Size Factor} \quad (3)$$

Table 2.2 Suggested Rain Garden Size Factor in rain garden (Bannerman & Considine, 2003)

	Rain gardens less than 10 m from downspouts			**Rain garden more than 10 m from downspouts**
Depth of rain garden depression	10-15 cm	15-18 cm	20 cm	All depths
Sandy soil	0.19	0.15	0.08	0.03
Silty soil	0.34	0.25	0.16	0.06
Clay soil	0.43	0.32	0.20	0.10

This sizing method is based on the ideal consideration that the rain garden would absorb 100% runoff from up to an inch of rainfall diverted from the total drainage area. This method is generally reliable. However, the main concern of this method is that the determining of the ponding depth is only based on the site slope is controversial while the given range of ponding depth is rather limited, while there was no scientific evidence to prove the effectiveness of the Rain Garden Size Factor.

There is another simple but effective way to determine the sizes of rain gardens suggested by Woelfle-Erskine and Uncapher (2012) and is also widely accepted in a variety of North American technical manuals such as 'A Rain Garden How-To Manual for Jeffersonville Homeowners' (Jeffersonville's Green Infrastructure Initiative, 2011). In this method, rain garden size is calculated by using the following equations (4) (5) (Woelfle-Erskine & Uncapher, 2012).

$$\text{Rain garden area} = \text{Runoff volume} \div \text{Ponding depth} \quad (4)$$

$$\text{Runoff volume} = \text{Total drainage area} \times \text{Design rainfall intensity} \times \text{Design duration of rainfall} \quad (5)$$

In this method, the total drainage area is calculated by applying the same method in the previously stated method. The ponding depth is effectively determined from the on-site soil infiltration rate (i.e. the rate at which rainwater can be absorbed into soil) and the designed dewater time (i.e. the designed time to completely drain the retained water in the rain garden) (6). As stated previously, many North American rain garden manuals suggest rain gardens to completely dewater within 24 to 96 hours (Bannerman

& Considine, 2003).

$$\text{Ponding depth} = \text{Soil infiltration rate} \times \text{Designed dewater time} \quad (6)$$

2.6 Plantings in rain gardens and the image of taxonomically diverse communities

As noted previously, vegetation plays a major role in the efficiency and success of the quantity and quality treatment of urban stormwater. Plantings can also greatly add various ecological benefits and aesthetic value to the urban living environment while the sustainable stormwater management components are potentially widespread in the contemporary green infrastructure (GI), especially of the rain gardens (Steiner & Domm, 2012). Plant types intended for use in rain gardens can range from flowering forbs, ornamental grasses, trees and shrubs, which are determined mainly by geographic area, climate, sizes of rain garden depressions and their aesthetics that appeal to people. Flowering forbs (especially of the herbaceous perennials) and ornamental grasses are capable for use in rain gardens at any scale, and are most popular plants adopted in rain gardens for their visual appeals with the variation in their forms, flower colours, blooming periods, and foliage textures. Specimen trees are preferred in rain gardens to add seasonal interest with their forms, flowers and colour of foliage and barks (Fig. 2.21). Shrubs can be used to provide structure, colour (e.g. evergreen shrubs with year-round evergreen leaves, deciduous shrubs with additional leaf colour at different times throughout the growing season, blossoms or colourful winter berries, etc.) and larger focal points for visual interest in rain gardens (Fig. 2.18) (Woelfle-Erskine & Uncapher, 2012). Rain gardens employ plants with colours and textures that appeal to people. Mixes of diverse species with colourful flowers and foliage (e.g. border-like plantings, Fig. 2.22) and monoculture or mown vegetation with low species richness (Fig. 2.23) are commonly seen in contemporary rain gardens.

Fig. 2.21. Specimen Birch tree adds aesthetic appeal to a domestic rain garden with its graceful form and interest of creamy-white barks. Picture source: http://www.sidneyetienne.com/wp-content/uploads/2015/05/plants-for-a-rain-garden.jpg.

Fig. 2.22. Border-like plantings featuring a mix of ornamental grasses and colourful flowers brighten up rain gardens over time, Sheffield, UK. Photo was taken by the author in July 2013.

Fig. 2.23. Municipal rain gardens featuring plantings with low species richness but tidy appearances, Columbus, USA. Photo was taken by the author in October 2012.

When bringing in considerations of strengthening the ecosystem services and aesthetic performances of vegetation in rain gardens, many technical guidelines suggest planting a diverse array of species in these features (Atchison et al., 2006; Dunnett & Clayden, 2007; Steiner & Domm, 2012). These designed taxonomically diverse plantings often feature the optimum plant communities that consist of a variety of plants that not only thrive in the indigenous conditions but also happily coexist together (Kingsbury, 1996). The intention of the taxonomically diverse plantings is to mimic natural plant communities that have the wilderness characteristics and high species richness. The most valuable starting point to deliver taxonomically diverse plantings for better amenity and functionality in rain gardens is to seek inspiration from nature. There are many potentially suitable plant communities from natural habitats that are similar to the rain garden conditions.

Meadows and prairies are two widely adapted examples of reference communities to provide the desirable dynamic flora and colourful floweriness for rain gardens (Dunnett and Clayden, 2007; Steiner & Domm, 2012). According to plant ecologists, the term ‘meadow’ refers to the spontaneous cool-season forbs and grasses on ground beyond the boundary of the tree canopy (Hitchmough, 2003; Lloyd et al., 2004), while the term ‘prairie’ is defined as the natural grasslands dominated by warm-season grasses and forbs that is regionally restricted to North America (Lloyd et al., 2004). Typical wildflowers in most meadows and prairies grow best in open swards where their competitive bedfellows (e.g. grasses, scrubs and woods) are sparse (Lloyd et al., 2004). It is thus noticeable that the beauty provided by wildflowers in the two reference communities respond to the interaction they have with their growing environment. For example, most British meadows exist because man cleared woodlands to have the lands for stock grazing or haying, in which the consistent man-management continued to evolve into habitats where wildflowers could colonise without immediate undue competition from grasses and scrubs (Lewis, 2003). Fire has a fundamental role for prairies, while fire disturbance can not only create the abundant open field for a wide suite of prairie-dependent flora, but also suppress a number of invasive plants, as well as to increase germination and flowering of native wildflowers (Martin et al., 2014).

Meadows and prairies can be florally diverse. As stated previously, the forage management practices (e.g. grazing and hay cutting) and fire disturbances can create the ever-richening earth and reduce the abundance of competitors for the colonisation of more wildflower species to increase plant diversity (Questad et al., 2011). Various different species rise and in their abundance over time in meadows and prairies to set up the vegetation succession, which may be a result of the different length of lifecycles in different species, or a result of the outcome of between-species competition (Dunnett, 2003). Succession within the species mixes allows species replacement to occur, not only insures the continuity of the integrity of meadows and prairies (Dunnett, 2003), which is also another fundamental driver leading to greater community dynamics in the two naturalistic reference communities including greater species diversity, structural diversity and changes in landscape structure (Waldhardt & Otte, 2003).

When in their natural habitats, meadows would normally occur in places where are typically humid in the summer and may thus require ample moisture (Druse & Roach, 1994). Therefore, meadows might be a good reference community to find constructive species that are well-suited to the constant moist soils in rain gardens. For example, spontaneous meadows in damp wild areas at Stilligarry, UK, could be found at their peak flowering display in June and July, dominated by *Trifolium pratense* and *Galium verum* (Fig. 2.24) (Gibbons, 2014). The Western Yunnan alpine meadow in Shangri-La Region, China, has the rich temperate flora providing splendid floweriness between June and September with plenty of rainfall at times (Gibbons, 2014), while many of the species such as *Primula secundiflora* and *Iris chrysographes* are assumed to be good plant options in rain gardens for their visual appeal and the ability to withstand periodic inundations (Fig. 2.25). However, it is also worth mentioning that meadows may not always be moist. For example, meadow communities may be capable of thriving the semiarid lands in boreal or Mediterranean climate, but could not persist for long without grazing or haying due to natural reforestation (Coiffait-Gombault et al., 2010; Yakimenko, 1996). There are a number of meadow species that can tolerate a wide range of moisture conditions, such as the *Iris sibirica* that occurs in wet European native meadows which may also thrive in prolonged drought (Hansen & Stahl, 1993).

Fig. 2.24. British native meadows grown in indigenous damp areas at Stilligarry, UK, with the two predominant species of *Trifolium pratense* and *Galium verum*. Picture source: http://images.fineartamerica.com/images-medium-large/8-wildflower-meadow-bob-gibbons.jpg.

Fig. 2.25. Spontaneous alpine meadow occurring in damp lowland, dominated by *Primula secundiflora* and *Iris chrysographes*, Shangri-La, China. Photo was taken by the author in July 2013.

Prairies often occur in places that have warmer temperatures and less precipitation than where meadows hail from (Druse & Roach, 1994), so that drought tolerant species to strengthen rain garden vegetation in prolonged drought might be provided from prairies. For instance, *Solidago Canadensis* and *Rudbeckia fulgida* are two relatively drought tolerant species naturally occurring in North American prairies (Fig. 2.26), and are widely adopted in US rain gardens for adding drought tolerance in plant communities and brightening up the gardens with their showy yellow flowers from late summer to autumn. It is also noticeable that there are also a number of the drought-tolerant prairie species may thrive in moist to damp soils, such as *Aster laevis* and *Mondarda fistulosa* (Hitchmough, 2003).

Fig. 2.26. Spontaneous North American prairie dominated by *Solidago Canadensis* and *Rudbeckia fulgida*, Columbus, USA. Photo was taken by the author in October 2012.

When meadows and prairies are adapted in urban vegetation, such communities are capable of thriving abiotic factors in urban landscape and are attractive to citizens, while extensive management (e.g. cutting back for a couple of times in a year) is necessary for their longevity (Hitchmough & Wagner, 2011). When fall within the scope of planting design, these two terms are likely to be described as the artificial random mix of herbaceous species, in which the flowery forb species are often adapted as the major component for providing aesthetic appeal that mimic the reference communities from natural habitats (Hitchmough, 2003). The urban artificial meadow communities can be visually distinguished from the artificial prairies as they may adopt many European/Asian indigenous forb species while the later would mainly adopt North American species as the community dominants. All constituent species from the random mix of meadows and prairies shall be delicately selected to provide urban visitors with a pleasant display of an area that is rich in a wide variety of wild herbaceous flowers in their blooming seasons. The seasonal flowering displays of such mixes are normally dominated by several species that are either bigger in biomass or have flowers with large sizes or significant colours, while the rest of the species coexist with the dominant display species in order to have either a harmonious or sharply contrasting mix of colour (Hitchmough, 2003; Lloyd et al., 2004). For example, Fig. 2.27 shows the flowering display of a rain garden created from in-situ sowing British native meadows in June 2015, which was dominated by *Leucanthemum vulgare* with visible large white flowers, while the pink flowers of *Lychnis flos-cuculi* and golden flowers of *Ranunculus muricatus* provide a complementary sharply contrasting colour mix. The community of such taxonomically diverse planting is normally complex with different plant traits, different shapes and textures in stems and leaves and various heights in plants (Kingsbury, 1996).

Fig. 2.27. Rain gardens featuring European native wildflower meadows, Sheffield, UK. Photo was taken by the author in June 2015.

Designed taxonomically diverse planting inspired by the reference communities such as meadows and prairies is attractive for raising public acceptance (Nassauer, 2004; Wagner, 2008), which emphasises a rich combination of various colours, textures and forms from the use of various plants, and the fascinating phenological display over time derived from the use of more species which cover different bloom periods in a year (Kaplan & Kaplan, 1989). Taxonomically diverse plantings can interact with local biodiversity by providing habitats to support many flower species and especially those beneficial but rare species that can hardly grow elsewhere (Rodwell et al., 1992; Lloyd, 2004). Such plant communities can then provide food (e.g. nectar and seeds, etc.), habitats and shelters to support a wide range of invertebrates such as butterflies and beetles, along with birds and mammals for different stages of their life cycles (Dunnett & Clayden, 2007; Lloyd, 2004).

Taxonomically diverse plantings have been adopted in urban areas in European and North American regions since the 1980s, and have become popular as habitat restoration practices to counteract the dramatic loss of biodiversity in cities (Marshall & Moonen, 2002; McDonald, 1993). Norderhaug et al. (2000) figured out that the fragmentation of meadows could result in a profound decrease in local biodiversity. Therefore, there is a growing interest in research on the establishment and management of taxonomically diverse plantings such as meadows and prairies (Chapman *et al*., 1996). For instance, Hitchmough (2000) carried out a 3-year field experiment to monitor the establishment of twenty cultivated herbaceous perennial species by planting into a sown native wildflower meadow in Ayr, South-west

Scotland. This study has particularly looked into the interactive competition between the selected species by planting and the wildflower meadow communities, where most declined as a result of competition and further suggestions were given for providing attractive taxonomically diverse flowery plant communities in urban parks (Hitchmough, 2000). Zechmeistera et al. (2003) have shown that increasing the intensity of mowing and application of fertilizer has negative impacts on plant diversity of wildflower meadows. Oudolf and Kingsbury (2013) suggested the use of some of the most popular, easily propagated and resilient plants that nonetheless have poor aesthetic values after flowering or growing season (e.g. plants with absence of flowers and leaves and dead tissues break down as debris to make unpleasant appearances in landscape, Fig. 2.28) should be less than 30% in a designed taxonomically diverse planting, so that people would prefer a persist and flourish planting all year around rather than give too many inputs for the maintenance after the flowering periods (e.g. re-planting and cutting back). There are a number of key documents for the introduction of ecological benefits, plant selection instructions and designing of taxonomically diverse plantings in urban areas, which include but are not limited to ‘Meadows’ (Lloyd et al., 2004), ‘The Illustrated Wild Flower Finder's Calendar’ (Lang, 2001), ‘Planting: a new perspective’ (Oudolf & Kingsbury, 2013), ‘New Perennial Garden’ (Kingsbury, 1996), ‘The natural habitat garden’ (Druse & Roach, 1994) and ‘Planting design: garden in time and space’ (Oudolf & Kingsbury, 2005).

Fig. 2.28. Planting remaining debris and untidy dead tissues outside of their growing season in bio-swale, Sheffield, UK. Photo was taken by the author in February 2013.

2.7 The contemporary issues in planting of rain gardens

As noted previously, taxonomically diverse plantings such as wildflower meadows and prairies are highly recommended for rain gardens. However, the use of such plantings in contemporary rain garden settings is currently at a 'learning-by-doing' stage, that not many rain gardens have adopted successful examples of such plantings. In fact, the importance of the taxonomically diverse plantings has often been underestimated, and the lack of plant diversity and inappropriate plant selection due to inadequate knowledge of proper vegetation was found to greatly influence the design of rain gardens as well as other sustainable stormwater management components (Shaw & Schmidt, 2003; Gilroy & McCuen, 2009). Moreover, wasteful implementation (e.g. seasonal beddings and transplanting pot plants grown from greenhouses, etc.), and the conventional maintenance in plantings (e.g. intensive mowing and the use of herbicides, etc.) were, and still are, widely applied in rain gardens. These conventional techniques may negatively impact the stormwater treatment in rain gardens and are potentially harmful to environment (Steiner & Domm, 2012, Yang et al., 2013). Details in all these concerns and the research gaps in contemporary plantings in rain gardens will be critically reviewed from a big picture.

2.7.1 Planting diversity of rain gardens

Planting diversity may help sustain the stability and long-term aesthetics of the plant communities and interact with local biodiversity. According to the principle that increased biodiversity stabilises community and ecosystem processes (MacArthur, 1955; May, 1973; Odum & Barret, 2004, Wilson, 2010), increasing the species diversity of urban planting may be one of the approaches to allow the landscape ecosystem to perform ecosystem services more effectively (Beck, 2013; Isbell et al., 2011). Increased biodiversity may result in increased resource-use efficiency (Tilman et al., 1996) and promote the development of self-sustaining urban plant communities that are stable over time in the face of environment change (Fontaine et al., 2006; Loreau et al., 2001; McCann, 2000). Planting diversity can sustain local biodiversity (Potts et al., 2005). In return, the enhanced biodiversity will contribute to the long-term sustainability and stability of the plant community with the strengthened key ecosystem services such as pollination for plants' sexual reproduction and colonisation with the migration of birds and small animals (Steffan-Dewenter & Tscharntke, 1997; Vickery et al., 2001; Menz et al., 2011).

Taxonomically diverse planting such as wildflower meadows and prairies compared to the conventional monoculture and mown vegetation has higher species richness. As noted previously, adaptation of these plantings in rain gardens can provide more available plants over time for providing the various environmental benefits as stated previously, but also maintaining the vegetation's service of stormwater management (Dunnett et al., 2008; Johnston, 2011), as well as adding aesthetic amenity (Steiner & Domm, 2012). Furthermore, the loss of natural habitats due to the expense of green spaces in urban areas lead to a significant increase of biodiversity hotspots that contains numerous endemic plant species and herbivore guilds (Myers et al., 2000; Beck, 2013). Rain gardens can work as a potential urban germplasm bank to facilitate the restoration efforts, which provide ecological opportunities to collect the beneficial and endangered plants as much as possible and maintain them in the persistent taxonomically diverse plantings.

However, as stated previously, contemporary urban green infrastructures are often highly managed, in which the lack of plant diversity is the major concern that threatens the biodiversity conservation, longevity, and amenity of the systems (Dunnett & Clayden, 2007; Grimm et al., 2008; Lovell & Taylor, 2013). Similarly, rain gardens have often been engineered without considering the vegetation possibilities, in which monoculture of single plant species or vegetation composed with extremely low species richness are commonly seen (Fig. 2.29). These plantings have been consistently criticised for their monotonous appearance (Shields, 1990; White & Gatersleben, 2011). The lack of diversity in vegetation may lead to unnatural and sometimes unpleasing visual performances of plantings that do not have sufficient species richness to reveal the beauty of phenological changes to sustain visual interest over time (Dunnett, 2004). Plant communities with a monoculture or a low genetic variation may lead to the severe impact on their population survival due to the failure of a single or more species caused by climatic extremes, disease or pests, which are thus associated with a high maintenance requirement (Chapin III et al., 2000; Beck, 2013; Holling & Meffe, 1996). Nevertheless, monocultures or compositions with extremely low species richness may only provide minimal habitat value (Dunnett & Clayden, 2007). All the concerns in terms of planting diversity in rain gardens result in an important research gap. There is therefore an urgent need for the involvement by horticulturists and consideration of the wider potential for planting these features.

Fig. 2.29. Rain garden features monoculture of single plant species, Columbus, US. Photo was taken by the author in October 2012.

2.7.2 Plant selection for rain gardens

Vegetation health is vital to the delivery of their ecosystem services, and is therefore one of the dominant factors to contribute to the success of rain gardens. Vegetation health in rain gardens greatly depends on the suitable plant selection to tolerate the environment stress in specific rain garden conditions. Moisture is the vital factor affecting the environment in rain gardens. As noted previously, most rain gardens are designed to have accelerated infiltration and evapotranspiration to avoid extended periods of waterlogging, so that they would only temporarily retain stormwater. Moisture condition in rain gardens periodically swings between the 'wet-mesic-dry' modes. Cyclic flooding is a distinctive condition in rain gardens, which refers to the repeated stages of flooding and draining over time (Dylewski et al., 2011). Typical rain gardens rely on precipitation as their source of irrigation, drought conditions are therefore expected during dry periods (Dunnett & Clayden, 2007). Rain gardens in most temperate climates are very unlikely to completely dry out for prolonged periods as there will be some soil moisture reserve at lower levels in its ponding shallow depression, and similarly, such features would only be wet during or immediately after a rainfall event and excess water will be progressively drained away (Dunnett & Clayden, 2007). Furthermore, there is a gradient of moisture levels in a typical rain garden depression structure. The bottom of its depression often has consistent moisture and receives the most water and does the majority of the water infiltration. Side-slope between margin and bottom often has moderate moisture that is greater than the moisture in the margin but reducing from the depression bottom towards to the margin of the depression. The rain garden margin and upland area beyond the margin are hardly waterlogged, and thus are often the driest zones in a rain garden.

The modern plantings in rain gardens may sometimes assemble plants from a variety of habitats so long as they are attractive to city inhabitants, without how the plants behave ecologically in their indigenous conditions, and the habitat demands for the completion of the life cycle of certain plants (Dunnett & Clayden, 2007). Inappropriate plant choices can result in failure of plant establishment in rain gardens due to their intolerance of the potential inundation conditions and extended drought events (Dunnett & Clayden, 2007; Dylewski et al., 2011; Shaw & Schmidt, 2003). Winning over public awareness is important for promoting the widespread use of rain gardens, as these features are relatively new in urban green infrastructure (Steiner & Domm, 2012). The health of plants is always a key component in public perception of success, as nobody would prefer the poor aesthetics of dead plants. Inappropriate plant choices could not contribute to an ecologically appropriate community, but may result in considerably increased maintenance works (e.g. seasonal bedding, replacement of failed plants, etc.) associated with undesirable waste of resources (Kingsbury, 1996) or undesirable invasive plants (Steiner & Domm, 2012). The failed planting in rain gardens would result in poor visual appearances and poor delivery of ecosystem services (Fig. 2.30) (Dunnett & Clayden, 2007). The lack of successful examples of specific planting choices for rain garden conditions also leads to some projects with bare soils in the side-slope and bottom of the depressions of rain gardens without any vegetative treatments (Fig. 2.31), despite the fact that these two saturation zones normally receive the most stormwater and perform the majority of the infiltration and evapotranspiration functions. Furthermore, altering the landscape environment to suit unsuitable plants, for example by the use of fertilizers to landscape soil and pumping groundwater for plants which require extended moisture in a water-stressed site, can be highly wasteful of resources (Hansen & Stahl, 1993).

Fig. 2.30. Planting failure caused by inappropriate plant selection in a series of highway rain gardens. The failed vegetated zone was occupied with native weeds. Xi’an China. Photo was taken by the author in June 2014.

Fig. 2.31. Public rain garden featuring dramatic plantings on only marginal areas without vegetation coverage on top of its side-slope and bottom, Sheffield, UK. Photo was taken by the author in June 2015.

In addition, there is a native/non-native debate that imposes a limiting factor on plant choices for rain gardens (Dunnett & Clayden, 2007). There is a long-term debate that all the species imported from outside regions have tended to be judged as unquestionably bad for their invasiveness, which would harshly threaten biodiversity and fragment habitat (Parker et al., 1999; Peretti, 1998; Schmitz & Simberloff, 1997) and only provide limited support for native fauna (Wilcove et al., 1998). To date, most rain garden guides are North American and Australian sources, where ecologies have been involved, vegetation is restricted to a native plant palette, as are always claimed to be less suitable for local environment and climate and highly invasive (Steiner & Domm, 2012). There is no doubt that invasive species should be controlled in rain gardens to prevent them outcompeting other beneficial local species. However, invasiveness is not a unique property of non-native species (Sagoff, 2005) as some of the native species with highly productive characteristics are hostile to other species or dominate their neighbours in a community (Pakeman & Marrs 1993; Brown, 1999). In fact, most cultivated species do not in any case possess the characteristics typical of invasive species (Thompson et al., 1995). Many plant species are proved to be adapted to local environment and climate as long as they are imported from similar habitat conditions as the indigenous growing conditions (Dunnett & Clayden, 2007). The debate for the limited habitat value of exotic species compared to the native species is also increasingly untenable. Smith et al. (2006) and Owen (2010) indicated that non-native plants do greatly support native fauna and provide ecosystem services in a similar way to native species in urban ecosystems (Hitchmough & Wagner, 2011) and even are culturally important to people (Hitchmough, 2011). Sustainable urban vegetation can be created by both native species and non-native species drawn from biogeographically similar regions (Hitchmough, 2011; Kingsbury, 2004). Unusual alien plants with dramatic appearances may be useful in winning public support for change. Climate change also encourages species migration (Cameron et al., 2012), as well as gradually shifts the traditional plant species diversity and floristic composition (Thuiller et al., 2005), so that there is an opportunity to introduce the cultivated non-invasive exotic flora from the appropriate reference climatic regions to create new urban sustainable plant communities.

However, vegetation options are barely studied and only limited reference plant communities are out there which could be found from the same habitat conditions (i.e. seasonal wetlands or boundaries of water bodies) under similar climates (Dunnett &

Clayden, 2007). Only little data-based research on the feasibilities of the suggested plant species in typical rain garden conditions are reported. The available scientific literatures are either testing rather limited diversity in plant traits in rain garden conditions, or failed to reflect the correct understanding of the hydrological conditions for rain gardens. For instance, Dylewski et al. (2011) flooded pot plants in controlled water bath for different durations and repeated the inundation-draining cycles to observe the effects of cyclic flooding on the survival and growth of only three candidate North-American native shrubs, where the performance of the other widely used plant traits such as ornamental forbs and grasses in rain gardens were not reported. Vander Veen (2014) visually judged the growth conditions of a series of preferred North American native trees, shrubs, forbs, ferns, grasses and vines for rain gardens in saturated situations and determined their stress in drought conditions from the on-site measured plant available water. However, this study failed to address the performance of plants in typical cyclic flooding conditions (i.e. the repetition of inundation and draining stages) but only draw conclusions on how the vegetative health of plants was affected by random inundations and droughts. Therefore, obtaining reliable data for the effects of typical rain garden conditions (e.g. cyclic flooding, moisture gradient throughout the depression and potential drought) on the growth of different plant traits from different regions is the main focus of this PhD study.

2.7.3 Sown taxonomically diverse plantings in rain gardens

Similar to the situations in urban plantings, seasonal changes of plants and transplanting seedlings or pot plants in rain gardens can greatly increase the budgets of installations for spending on labour, seedling cultivation in nurseries and transportation (Dunnett & Clayden, 2007). Plant cultivation in greenhouses may also result in the resource consumptions on irrigation, heating and cooling. To cope with these issues, Dunnett and Clayden (2007), Hitchmough and Wagner (2013) recommended sowing seed mixes in situ as the alternative approach to establish the taxonomically diverse plant communities in rain gardens. Compared with the traditional transplanting methods, sown plantings may take longer to produce the expected landscape but require smaller budgets and less input (Dunnett & Clayden, 2007), while sowing species-rich mixes could easily create a naturalistic display with great diversity on a large scale (Hitchmough & Wagner, 2013). A good example of the application of sown plantings in stormwater management components could be found in the 2012 London Olympic Park, where bioswales planted with sown British native forb-rich mixes were extensively used as an example of an ecologically informed landscape which also accelerated the infiltration of stormwater runoff collected from surrounding surfaces while providing visual appeal with great plant diversity and attractive flowering displays in summer time (Hitchmough & Wagner, 2013; Oudolf & Kingsbury, 2013). However, most sown plantings in rain gardens and other stormwater management components often adopt sown turf mixtures or monoculture of perennial grasses (Fig. 2.32) (Katuwal et al., 2008; Mazer et al., 2001), which are weak in aesthetics and habitat values, as well as in runoff reduction (Dunnett & Clayden, 2007; Steiner & Doom, 2012).

Fig. 2.32. Bioswale featuring sown turf mixes with low plant diversity and poor aesthetics, Sheffield, UK. Photo was taken by the author in May 2012.

Poor maintenance can turn rain gardens into unpleasant corners in urban public spaces (Fig. 2.33). However, conventional maintenance techniques may adversely alter the runoff management, aesthetics, and other ecosystem services in plantings in rain gardens. For instance, close-cropped mown vegetation in rain gardens may turn into a muddy quagmire thus resulting in a limited contribution to water management and local wildlife conservation (Steiner & Domm, 2012). The use of sown taxonomically diverse plant communities in rain gardens is expected to provide ecological opportunities to mitigate the impact of maintenance to urban environment, as these plantings are claimed to require less maintenance input such as minimal irrigation, fertilising and mowing (Dunnett & Clayden, 2007; Hitchmough & Wagner, 2013). However, a major concern in seeding rain gardens is that the urban soils often tend to carry a significant load of weeds in soil seed bank (Hitchmough et al., 2008), which could outcompete the desirable sown species especially of those relatively unproductive ornamental forbs (Hitchmough et al., 2008; Sluis, 2002) and thus adversely alter the landscape effect in rain gardens (Fig. 2.34). However, intensive weeding requires additional cost and labour, while some of the management means could be extremely harmful to urban environment. For example, conventional techniques such as the removal of topsoil with the embedded seed bank or using seed bank free soil substrates to restrict the emergence of weeds are wasteful and may also damage the future productivity potential of the soil (Hitchmough et al., 2008; Westbury & Dunnett, 2008). The use of herbicides for weed control could be particularly serious in rain gardens, as such measures often have hydrological connection between the public drainage systems or water bodies to lead the harmful runoff excess with the concentration of herbicides to downstream aquatic ecosystems (Yang et al., 2013). In practice, there are cost-efficient and less interventionist weed control means to contribute to minimise the use of herbicides and wasteful soil amendments, such as the mulch of weed-free substrates to give the sown species a head establishment and to build their advantages in weed competition (Dunnett & Nolan, 2004, Getter & Rowe, 2006), and the use of biological solutions for mitigating the weed competition such as increasing the sowing rate (Dunnett & Clayden, 2007) and the use of hemiparasitic plants (e.g. *Rhinanthus minor*. Westbury & Dunnett, 2008). However, the effectiveness of the simple weed control means remains unreported in sown seed mixes for rain gardens, which leaves a great research gap, and is thus another focus of this PhD study.

Fig. 2.33. Poorly managed vegetation in a street rain garden, Columbus, USA. Photo was taken by the author in October 2012.

(a) **(b)**

Fig. 2.34. An example of the weeded rain garden.

a: Rain garden featuring a sown planting dominated by *Iris sibirica* and *Lobelia cardinalis* in July 2013, Sheffield, UK. Photo was taken by the author in July 2013.

b: The sown community was outcompeted by British native weedy species including *Agrostis capillaris*, *Koeleria macrantha* and *Veronica chamaedrys* in July 2014, Sheffield, UK. Photo was taken by the author in July 2014.

Furthermore, there are rather limited studies that reflect the success of the sown species-rich mixes in rain gardens. Hitchmough and Wagner (2013) carried out a five-year study on the establishment of a sown mix of uncompetitive rosette-forming forbs intended to be used in rain garden vegetation subjected to simple management treatments (i.e. irrigation and cutting in September and November). In this study, the long-term persistence of many species was improved on soils with more moisture content (Hitchmough & Wagner, 2013). The concern of this study is that the responses in these sown plantings towards the 'wet-dry' moisture gradient throughout the "margin-slope-bottom" structure in practical rain gardens were not explained. The unreported interaction of proposed seed mixes and the stated different saturation zones (e.g. the dry margin, moderate moist slope and damp bottom of the rain garden depression) is also one of the research objectives of this PhD study.

2.8 Review of rain garden success in urban stormwater management and urban biodiversity

2.8.1 Hydrological performances of rain gardens

Rain gardens largely employ vegetation and soils to replace the urban impervious surfaces and to replicate the ecological processes of runoff reduction in natural systems (Steiner & Domm, 2012). The retained water progressively receives treatments with vegetation and soils before the excess water is released into public sewage or drainage systems (Dietz, 2007). With the vital ecological processes including retention (i.e. retaining runoff in their shallow depressions), infiltration (i.e. the downward movement of rainwater through soil and bedrock) and evapotranspiration (i.e. the combined efforts to the water loss from evaporation from soil surfaces and transpiration of runoff back into the atmosphere via vegetation) that are improved by the physical and biological properties of vegetation and soils, the quantity of stormwater are expected to be reduced within rain gardens (Steiner & Domm, 2012; Coffman & Winogradoff, 2002).

Runoff reduction in the rain garden systems is one of the most active aspects in current research of sustainable stormwater management. For instance, a North American study of Bedan and Clausen (2009) demonstrated that a significant runoff reduction could be derived from the application of sustainable stormwater management components. In this study, runoff quantity from a traditional residential design (impervious roads and grassed lawns draining directly to public sewer systems) and a new development installed with rain gardens and grass swales were measured weekly at the approved discharge pipes during the predevelopment, construction and post-construction period of the establishment of the traditional development and the new development installed with rain garden and bioswale settings (Bedan & Clausen, 2009). Bedan and Clausen (2009) concluded that the mean value of the amount of runoff per week was increased by 16 times during the post-construction period (19 June 2003 to 30 June 2005) from the traditional developments compared to predevelopment (4 April 1996 to 8 October 1997), while the mean runoff volume per week was reduced by 42% during the post-construction period (18 January 1996 to 23 March 1999) within the watershed retrofitted with rain gardens and bioswales as compared to predevelopment (1 August 2002 to 30 June 2005). Yang et al. (2013) measured the runoff outflows from two rain garden models planted with six North American native plant species under natural storm events in Ohio State, USA. The experimental rain gardens reduced significant amounts of runoff for 59% in the rainfall events between 6 and 12 mm, and for 54% in the rainfall events more than 12 mm (Yang et al., 2013). Significant reduction in peak runoff flow rate (i.e. the greatest runoff velocity during a rainfall event) was also found for 84% and 88% in the rainfall events between 6 and 12 mm, and that of more than 12 mm, respectively (Yang et al., 2013).

Runoff reduction occurs in rain garden systems because of the effects of interception and infiltration (Dunnett & Clayden, 2007). Vegetation can intercept precipitation with their canopies and reduce the velocity at which the rainfall falls onto the ground, thus prevent stormwater from rushing over the urban surfaces (Xiao & McPherson, 2002). Interception pattern derived from vegetation largely relates to the structure of the canopy traits (Clark, 1937; Clark, 1940). For instance, taller and widespread plants with larger canopies that cover a larger surface area can substantially decrease the amount of the stem flow (the amount of runoff entering the soil at the base of trunks) and throughfall (the amount of rainfall coming through the canopy) (Fig. 2.35) (Anderson et al., 1969; Ford & Deans, 1978; Gilliam et al., 1987;

Thurow et al., 1987). It also leads to a hypothesis that vegetation with greater structural complexity in their aboveground traits may provide better rainfall interception than that of relatively uniform structural diversity.

Stormwater beyond the excess of soil field capacity (i.e. the amount of soil moisture content held in a soil after the free drainage of excess water has ceased. Israelsen & West, 1922) can be intercepted and temporarily retained in the depression structure of the rain garden systems until it reaches the storage limitation other than be diverted straight away into drainage systems. A portion of runoff that runs through rain gardens and can be absorbed into pore spaces in soils or be taken up and temporarily stored in plants, so that the runoff volume is reduced (Bengtsson et al., 2005; Dunnett & Clayden, 2007). Infiltration was found to be another dominant factor for the runoff reduction in rain gardens (Davis et al., 2001; Yang et al, 2009). Infiltration can be greatly hindered by impervious surfaces and compacted soils, while vegetation's root and decomposition can penetrate the compacted soil and create macropores thus greatly increasing the infiltration rate and maintaining infiltration capacity over time (Ball *et al*, 2005; Bartens et al., 2008; McCallum *et al*, 2004). Typical rain gardens may also employ soil amendments for increasing the soil porosity to improve the soil infiltration and the absorption of rainfall on site (Dietz & Clausen, 2006). Another benefit of infiltration is to recharge groundwater and maintain flow regime, as well as provide baseflow to nearby watercourse (Klein, 1979).

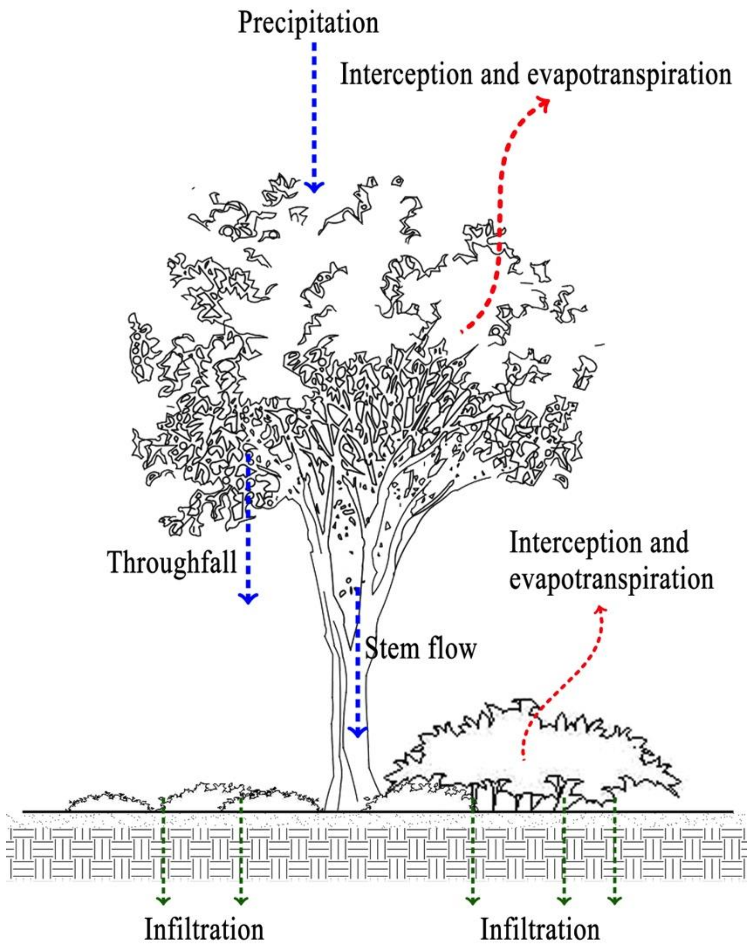


Fig. 2.35. Influences of stormwater management from vegetation and soils (adapted from Dunnett & Clayden, 2007).

In the water cycle restored by rain gardens and other sustainable stormwater management facilities, a large sum of the stormwater apart from the infiltrated part will return to the atmosphere via evapotranspiration from plants and soil surfaces so that the total amount of runoff is reduced (Fig. 2.35) (Grimmond & Oke, 1991; Mitchell et al., 2008). For instance, Hickman (2011) installed two rain gardens in Villanova, USA, for obtaining the on-site reduction of stormwater. In this study, both rain gardens had 15 cm ponding depth and were planted with North American native perennials and shrubs (Hickman, 2011). One of the rain gardens was backfilled with

76 cm depth of artificial soil mixed by 65% sands, 33% silts and clays and 2% organics and allowing free infiltration without the installation of underdrain pipes, while the other rain garden was backfilled with substrates mixed by 88% sands, 8% silts and clays and 4% organics to a depth of 40 cm and a 36 cm depth of a water storage layer underneath the substrate layer which was sealed by impervious liners to collected water for plants to draw on during dry periods (Hickman, 2011). During March and December 2010, the on-site precipitation and evapotranspiration from the two rain gardens were instrumented, Hickman (2011) concluded that the evapotranspiration from the free infiltrating design and the model with impervious water storage zone had contributed to 35% and 63% of the losses of total amount of stormwater (i.e. retention), respectively. This result suggested that infiltration would play the major role in reducing runoff in rain gardens, while evapotranspiration would dominate the runoff reduction when natural infiltration is eliminated by engineered structures.

Evapotranspiration is the water loss derived from evaporation from surfaces and transpiration through the vegetated surfaces via plant stomata in foliage (Allen et al., 1998). It is governed by evaporation from bare soils, and would be greatly increased in vegetated areas through the contribution of transpiration from plants (Allen et al., 1998). Heat is the vital energy needed for evaporation, so that the evapotranspiration in rain gardens and other sustainable stormwater management components can be greatly affected by climate, where the evapotranspiration would be reduced in cooler climates (Hunt et al., 2006; Poë et al., 2015). Previous studies found that the growth of vegetation canopy diameter (Vertessy et al., 1995) and biomass (Nagase & Dunnett, 2012) may vary the transpiration though plants differ seasonally. Therefore, the reduction of stormwater runoff in rain gardens is expected to be higher via accelerated loss of water back to the atmosphere caused by increased evapotranspiration, which is improved by plants in growing seasons (Lundholm et al., 2010). Evapotranspiration in rain gardens may also be affected by their sizes, as larger shallow planted depressions may encourage maximum evapotranspiration for increasing runoff reduction (Dunnett & Clayden, 2007). However, in practice, the contributions between the two simultaneous processes of evaporation and transpiration to the combined term of evapotranspiration could hardly be distinguished from each other (Ward & Trimble, 2004).

Recent efforts to improve the efficiency of stormwater runoff reduction in rain gardens have primarily focused on the development of construction and size (Dussaillant et al., 2004; Le Coustumer et al., 2012), amending soil media compositions for effective runoff infiltration (Thompson et al., 2008; Carpenter & Hallam, 2010), which are predominantly normative from the engineering point of view. For instance, sizing of rain gardens has been emphasised by Brown and Hunt (2011) who indicate that better reduction in runoff amount could be seen with increasing ponding area and deeper media depth. Soil amendment is strongly recommended for a higher infiltration rate in the rain garden settings to avoid the soil medium prone to compaction and poor water movement, thus reducing the runoff overflow from these features (Cho et al., 2009).

As noted previously, a number of studies have examined the effectiveness of rain gardens to reduce stormwater runoff, in which vegetation is considered as one of the key functional factors. However, the development of planting design for increasing runoff reduction in rain gardens has not been successfully reflected in previous studies. In fact, the perceived contributions of vegetation to the change in hydrology of rain gardens were not fully revealed (Johnston, 2011). Rather limited experimentally replicated studies published in the world have specifically acknowledged the role of vegetation in the stormwater runoff reduction in rain gardens. For instance, a North American field rain garden experiment of Johnston (2011) indicated the role of vegetation in altering drainage dynamics in rain gardens through the changes in antecedent soil water, plant-induced differences in soil structure (e.g. increasing soil porosity to improve the infiltration rate) and the evapotranspiration that ultimately led to the reduction of the percentage of runoff output volume. Nagase and Dunnett (2012) tested a range of plant types including grasses, forbs and sedums in models backfilled with vegetation and commercial substrates. In this study, vegetated models were treated with two different artificial rainfall events (100 mm/h and 50 mm/h) in a controlled environment in which the temperature was kept at more than 20°C, and the results demonstrated that different plant traits and the size of plants, as well as vegetation biomass could significantly affect the runoff discharge from the systems. However, previous studies tend to show highly variable observations due to different vegetation choices or experimental conditions. For example, Davis et al. (2001) reported an indoor experiment that two rain garden models planted with 24 creeping Junipers, which were adopted in impervious containers (each had a dimension of 305 cm (length) × 152 cm (width) and 91 cm (depth)) and discharge the excess runoff from pipes installed at the bottom of the systems. In this study, the experimental models were given synthetic rainfall at 4.1 cm/h for 6 hours, and the results suggested a rather low evapotranspiration from the systems which only reduced 1% of the total volume of precipitation. Such results are not surprising as the small-leaved and slow-growing Juniper does not transpire much compared to many other higher plants. For instance, Selbig and Balster (2010) studied a series of practical rain gardens planted with turfgrass and prairies in Dane County, Wisconsin, USA, and suggested a much greater evapotranspiration from the vegetation than stated in the study of Davis et al. (2001), which could evaporate and transpire half of the annual rainfall input.

Furthermore, there are limited quantitative studies looking at the impacts of the taxonomically diverse communities to the reduction of stormwater runoff in rain gardens, where monoculture of trees, shrubs and herbaceous species are studied rather than as combined plant communities (e.g. Dylewski et al., 2011; Johnston, 2011). People may doubt the contributions of the taxonomically diverse plantings that are intended for use in rain gardens, due to the evident lack of reports for their hydrological performances in runoff reduction, which is potentially affecting the adoption of such plantings in these features. Only a few studies reported the comparison of the stormwater runoff reduction from the taxonomically diverse plantings and traditional urban vegetation such as mown grasses, but suggested highly variable results as experimental observations were restricted by test conditions (Johnston, 2011; Lundholm et al., 2010; Nagase & Dunnett, 2012). The remaining unclear effects of vegetation types, especially for the comparison between the suggested taxonomically diverse plant communities and the traditional vegetation such as mown grasses on the reduction of stormwater runoff in rain gardens leave important research gaps. There is therefore an increasing need to gather runoff quantity data from the different vegetation types in experimental rain gardens, which is also a main focus of this PhD study.

2.8.2 Pollution control of rain gardens

Security of urban water cycles are rather sensitive to the toxic accumulation in aquatic ecosystems associated with polluted urban runoff (Davis, 2005). The potential value of rain gardens in response to the challenges of urban non-point pollution conveyed by stormwater runoff in cities has been recognised to reduce the cost and resource consumptions for water extraction (Davis et al., 2009; Dietz, 2007; Endreny & Collins, 2009; Ermilio, 2005). For example, Lloyd et al. (2001) indicated the great effectiveness of pollutant removal in experimental rain gardens with a 60%, 47%, 66% and 29% reduction in total suspended solids load, total phosphorus, soluble phosphorus and soluble nitrogen, respectively. Previous studies also demonstrated that rain gardens are effective in reducing concentrations of heavy metals carried in runoff (e.g. Cu, Pb, and Zn, etc.) (Dietz, 2007; Davis et al., 2009).

Plants and soils play an important role in rain gardens on minimising the polluted stormwater discharge to the environment, as well as reducing the bioaccumulation of toxins in aquatic ecosystems (van Roon, 2007). For instance, suspended solids and particles (e.g. dust and soil particles and other debris) carried by urban runoff are settled out or filtered when running through soil and fibrous base of plant trunks and roots, whilst the uptake of mineral nutrients and heavy metal contaminants are held in the plants and recycled through successive seasons of plant growth, death and decay (Read et al., 2008). Dissolved substances would bind to the surface of plant roots, soil particles and humus, while the chemicals and organic matters are broken down by soil microorganisms (Dunnett & Clayden, 2007). The degradation (i.e. soil organisms' services for breaking down the organic substances) could be promoted in the oxygen-rich conditions surrounding plant roots (Dunnett & Clayden, 2007). Pollutant removals derived from the use of vegetation could be improved based on ecological principles, for instance, plants with high growth rates may be particularly effective for temporarily storing mineral nutrients (Dunnett & Clayden, 2007).

However, the removal of runoff pollutants provided by rain gardens may vary widely and is not acceptable at times (Yang et al., 2013). The wide use of herbicides for controlling weedy species in these features may lead to the concentration of herbicides in urban runoff, which remains one of the major obstacles in the widespread adoption of rain gardens (Cho et al., 2009; Hsieh & Davis, 2005; Maurakami et al., 2008; Yang et al., 2013). There is a fact that the soil amendments for accelerated infiltration rates might contribute to a rather limited time for runoff pollution treatment thus considerably compromising the pollutant removal efficiency in rain gardens (Yang et al., 2009; Amado et al., 2012). Conversely, poorly drained soil would have a particularly long storage of runoff and is more likely to cause water excess in rain gardens if a large rainfall event occurs in a short duration (Cho et al., 2009). Yang et al. (2013) thus designed a biphasic rain garden system that used an impervious anaerobic (water saturated) zone and U-shaped reverse drainage pipes allowing first flush runoff to be retained for a longer period of time for the completion of bio-infiltration treatment of polluted runoff and also retained a significant amount of runoff. This biphasic system also had an underground aerobic (water unsaturated) zone to further decrease runoff flow rate for subsequent aerobic treatment and only discharge the treated water into the recharge zone (Yang et al., 2013).

2.8.3 Ecology and biodiversity of rain gardens

The increasingly limited open spaces for biodiversity in the rapidly-urbanising catchments resulted in major urban habitat threats including habitat loss, habitat degradation and habitat fragmentation (Glanznig, 1995; Hammer et al., 2012) thus biodiversity conservation should be urgently taken at different scales in cities (i.e.

regional, local, and macro scale) (Rookwood, 1995). As noted previously, rain gardens can be adapted to any surrounding situations in urban developments, and thus are often regarded as one of the best opportunities for providing habitat restorations and biodiversity conservations in cities (Dunnett & Clayden, 2007; Steiner & Domm, 2012). The key factors in urban biodiversity are the taxonomic diversity and spatial complexity of urban vegetation (Smith et al., 2005). Such taxonomically diverse plantings with complex spatial structures and high species richness (e.g. wildflower meadows and prairies) are strongly recommended for the rain garden settings (Dunnett & Clayden, 2007; Steiner & Domm, 2012). These purposed vegetation for the benefit of wildlife in rain gardens might deliver the contributions including: (a) providing the urban germplasm bank for suitable and valuable native plants or even non-invasive exotic plant communities (Scher & Thiery, 2005; Hammer et al., 2012), (b) capturing and cleaning the urban runoff to guarantee the supply security of clean water for the survival of wildlife (Dunnett & Clayden, 2007; Woelfle-Erskine & Uncapher, 2012), (c) providing shelters for wildlife species that may be sensitive to human disturbance and adverse weather (Kazemi et al., 2011; Williams et al., 2010), and (d) providing more foods for sustaining the population of associated guilds over time with high species richness (Pottsa, *et al*., 2006; Weiner, 2011).

However, there is very limited literature reporting the impacts on local biodiversity from the installation of rain gardens, while most previous biodiversity assessments were specific to non-water related systems (Kazemi et al., 2009). For example, Kazemi et al. (2009) conducted a biodiversity investigation on 12 rain garden settings in Melbourne during the summer of 2006-2007, in which the terrestrial invertebrates were captured as biodiversity indicators through pitfall traps. The experimental observations suggested that larger numbers of plant taxa and greater plant litter depth could significantly contribute to the biodiversity in rain gardens (Kazemi et al., 2009). Larger interior habitat area (i.e. the vegetated area from the depression margin towards the centre) was also reported to promote biodiversity in rain gardens (Kazemi et al., 2009). In a 2011 study (Kazemi et al., 2011) involving the same biodiversity investigation method in nine rain gardens and nine corresponding common lawns on flat grounds in Melbourne, greater species richness was found in rain gardens than in the traditional lawns on flat grounds. It might because the rain gardens may often adopt a larger number of flowering forbs compare to the traditional lawns which may attract more invertebrate visitors (Kazemi et al., 2011). Studies of Kazemi et al. also proved that the rain garden systems played a role as a form of ecotone, i.e. the transition area between biomes in terrestrial landscape. Ecotones are often recognised as being biologically richer than either habitat adjacent to them on either side (Goebel et al., 2003; Palmer & Mazzotti, 2004). The 'wet' component within the rain gardens such as the negative lateral slope resulted in increased vegetation yield and coverage to support the herbivorous guilds, which also increased habitat heterogeneity and change in pH along the moist gradient to enhance the biodiversity of invertebrates compared to that of green spaces on floor areas (Kazemi et al., 2011).

Chapter 3: Plant selection for resilient perennial species in typical rain garden conditions

3.1 Introduction

3.1.1 Requirements for the plant selection for typical rain gardens

Most terrestrial stormwater management facilities, such as rain gardens, retention basins and bio-swales, are planted depressions that use vegetation and growing media (which is often amended for better infiltration rate) to improve stormwater infiltration and evaporation. Runoff reduction in rain gardens is higher when infiltration and evapotranspiration are significantly improved by the use of vegetation (Lundholm et al., 2010). However, plant selection for rain gardens can be complicated, as dynamic spatiotemporal moisture distributions are expected in most rain gardens through the typical depression structure.

There is expected to be a gradient of moisture levels in typical rain gardens due to the effect of gravity and the timescale of flooding. A typical rain garden consists of three saturation zones: depression bottom, slope and margin (Fig. 3.1). The depression bottom often stores more water over the side slope and margin and has a consistent moist or waterlogged state for an extended period of time in the wet season, whilst the occasionally flooded side-slope between the wetter bottom and dry upland margin will have a moderate moisture status (Dunnett & Clayden, 2007) (Fig. 3.1).

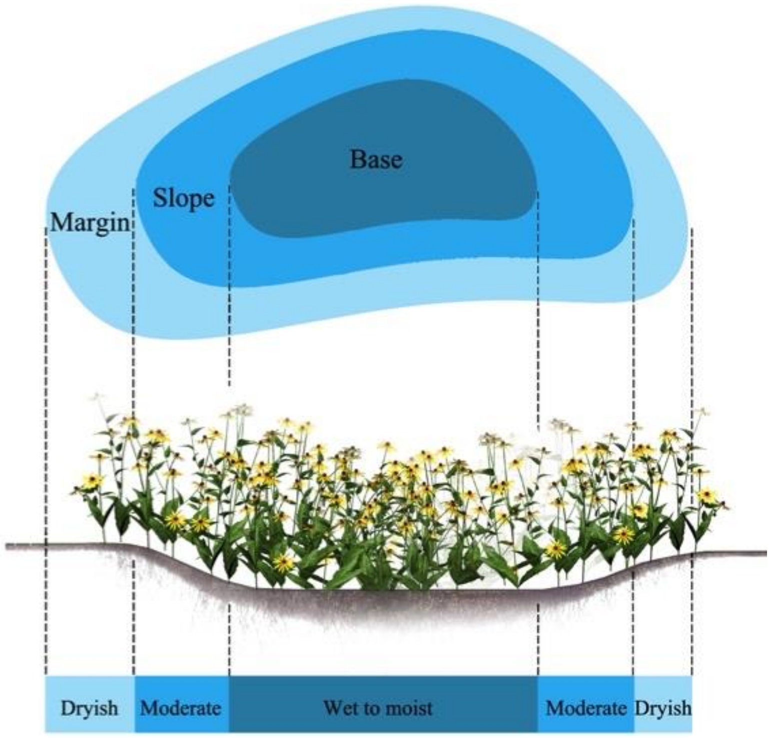


Fig. 3.1. Plan and section of the gradient of moisture levels in a typical rain garden

Cyclic flooding is expected to repeat over time in typical rain gardens, which consists of a cycle of waterlogging and draining (Dylewski et al., 2011). Dylewski et al. (2011) suggested that the rain garden soil might remain moderately moist for a few days after flooding, before drying out. Rain gardens normally rely on natural rainfall as their source of irrigation (Dunnett & Clayden, 2007), thus the rain garden vegetation may experience periodic drought conditions or even severe drought circumstances between precipitation events due to seasonal rainfall differences.

Rain gardens are normally specified to drain runoff rapidly to mitigate the risk of stormwater flooding (Cho et al., 2009). Appropriate specifications are given in many recent technical manuals to drain rain gardens from saturation within 24 hours, including soil amendments and the use of underdrain systems (e.g. BES, 2008; MDE, 2000; Prince George's County, 2001; Woelfle-Erskine & Uncapher, 2012). Dussaillant et al. (2005) suggested that the soil in rain gardens would likely remain saturated for one to two days, or much longer if they were sited on poorly drained clay or clay-loam soils. Therefore, the candidate plants for use in rain gardens should be able to withstand at least two days of waterlogged conditions. To avoid the risk of potential drowning, Davis et al. (2009) suggested that additional drainage facilities should be installed within rain gardens to ensure that the ponded water can be drained to empty within 72 to 96 hours for the designed rainfall event. Therefore, it is reasonable to design rain gardens to have the ability to dewater within a period from 24 hours to a maximum saturated period of 96 hours. Thus, suitable plants for rain gardens are ideally able to withstand this typical flooding cycle of saturated soil for between one and four days. In addition, considering the fact that rain gardens may experience potential dry circumstances between precipitations and the upper slopes of the depression may consistently remain dry, plant species that are capable of persisting during periods of limited rainfall or drought are also desirable for rain gardens and similar stormwater sustainable management components.

3.1.2 Plant selection for rain garden

Cyclical flooding leads to conditions in stormwater management facilities such as rain gardens that are similar to seasonal wetlands, with the added complication of the interaction with the gradient of moisture levels throughout the 'margin-slope-bottom' depression structure. Suitable vegetation types and plants for rain gardens are therefore hard to typify. To date, there are many plant recommendation lists available from technical manuals and documents. Such lists include, but are not limited to: Plant directory for rain gardens by moisture tolerance (Dunnett & Clayden, 2007; MCWCC, 2014); "Planting for rain garden" (Woelfle-Erskine & Uncapher, 2012); Plant lists organised by rain garden saturation zone and by sunlight requirements (Steiner & Domm, 2012); Wetland indicator status list (NJAES, 2011); Rain garden plants (Emanuel et al., 2010; Andruczyk et al., 2006); Plant lists organised by soil texture (Bannerman & Considine, 2003); Suggested plant lists for rain gardens in particular regions (e.g. Central Florida and Southeastern North Carolina, USA.) (D'Abreau, 2010; Glen, 2009); Native plant lists for rain garden designs (Metro Water Services Stormwater Department, Nashville, 2011); and grass choices for low-impact design (Lucas, 2011) etc.

According to these lists, a variety of plant types including herbaceous perennials and grasses, shrubs, specimen trees and ferns are recommended for use in rain gardens, while perennials and ornamental grasses are the two most common plant types in rain gardens. Some of the most commonly used perennial genera include: *Achillea* (e.g. Yarrow (*Achillea millefolium*)), *Aster* (e.g. Smooth Aster (*Aster laevis*), and New England Aster (*Aster novae-angliae*)), *Caltha* (e.g. Marsh marigold (*Caltha palustris*)), *Echinacea* (e.g. Purple coneflower (*Echinacea purpurea*)), *Hemerocallis* (e.g. Orange daylily (*Hemerocallis fulva*)), *Iris* (e.g. Siberian Iris (*Iris sibirica*) and Blue flag Iris (*Iris versicolor*)), *Potentilla* (e.g. Rough cinquefoil (*Potentilla norvegica*)), *Lilium* (e.g. Michigan lily (*Lilium michiganense*)), *Lobelia* (e.g. Cardinal flower (*Lobelia cardinalis*)), *Rudbeckia* (e.g. Black-eyed Susan (*Rudbeckia subtomentosa*), and Branching coneflower (*Rudbeckia triloba*)), Solidago (e.g. Ohio goldenrod (*Solidago ohioensis*), *Thalictrum* (e.g. Meadow rue (Thalictrum aquilegifolium)), and *Veronicastrum* (e.g. Culver's root (*Veronicastrum virginicum*), etc. For ornamental grasses, the genera including *Carex* (e.g. Tussock sedge (*Carex stricta*)), *Deschampsia* (e.g. Crinkled hairgrass (*Deschampsia flexuosa*) and Tufted hairgrass (*Deschampsia cespitosa*), *Juncus* (e.g. Sharp-flowered rush (*Juncus acutiflorus*)), *Miscanthus* (e.g. Japanese silver grass (*Miscanthus sinensis*)), *Molinia* (e.g. Moor grass (*Molinia caerulea*)), and *Panicum* (e.g. Switchgrass (*Panicum virgatum*)), etc., are given in most plant lists for rain gardens. These plants have gained popularity because of their visual aesthetics (e.g. colours, textures and variation in blooming periods, etc.), tolerance of a fairly wide range of hydrologic regimes and habitat values (Woelfle-Erskine & Uncapher, 2012), and as being capable for use for sites at any scale. Taxonomically diverse mixes of perennials and grasses are recommended in recent studies for their cost-efficiency in taking implementation and multiple functions including enhancement of stormwater infiltration and evaporation, aesthetic values and conservation of local biodiversity in stormwater management practices (Johnston, 2011; Hitchmough & Wagner, 2013; Nagase & Dunnett, 2012; Teemusk & Mander, 2007).

Ideally, perennials and grasses are established in planting positions appropriate to their ecological needs, resulting in lower maintenance and irrigation demands and greater longevity (Hansen & Stahl, 1993). It is sensible to select potential plants for rain gardens using species that originate from similar habitat conditions, such as seasonal wetlands, boundaries of water bodies, hay meadows or prairies that are subjected to periodic flooding (Dunnett & Clayden, 2007). Typical rain garden guides tend to propose suitable plant species on the basis of their assumed moisture sensitivities to different hydrological regimes. Moisture sensitivities of these plants tend to be determined according to their tolerance to fluctuation in flooding and drying documented in a variety of botanic guides for gardeners, such as the "RHS A–Z encyclopaedia of garden plants" (Brickwell, 2008), "Rain Gardens: Managing Water Sustainably in the Garden and Designed Landscape" (Dunnett & Clayden, 2007) and "Rain Gardens: Sustainable Landscaping for a Beautiful Yard and a Healthy World" (Steiner & Domm, 2012). Hydrological regime can be described by the duration, frequency, timing and predictability of the flooded and dry phases (Bunn et al., 1997). In general, four levels of moisture sensitivities are recognised, which range from: (1) continuous inundation (i.e. 'wetland' species), (2) periodic or seasonal inundation (i.e. species from wet meadows or other habitats that are not permanently wet), (3) infrequent inundation (i.e. species from fertile habitats in temperate maritime climates), and (4) intolerant of inundation (i.e. species from dry or arid habitats).

Current references of the plant choices in rain gardens often tend to recommend: (a) use species that can withstand the continuous flooding at the basin bottom in a poorly drained soil, (b) use plants that are capable of tolerating periodic to frequent inundations at the basin bottom in a well-drained soil, (c) use species that withstand infrequent flooding and seasonal dry spells at the side slopes, and (d) use plants that are widely available from dry and arid habitats at the most free draining and hardly flooded margins (Dunnett & Clayden, 2007; Steiner & Domm, 2012; Woelfle-Erskine & Uncapher, 2012). It is noticeable that the species assumed to withstand periodic or seasonal inundation and those assumed to tolerate infrequent inundation are the two most popular options in typical rain garden guides compared with the species inhabiting the other two hydrological regimes. Nevertheless, most of the plant lists are not based on data from replicated experiments, and there has been little research that evaluates the interaction between specific plants and the dynamic spatiotemporal moisture distribution in rain gardens. This leaves a major research gap in expanding plant options for rain gardens, and the appropriate plant selection method in typical rain garden conditions (i.e. cyclic flooding and potential drought, as well as the gradient of moisture levels) is not yet clearly addressed.

Most references advise the use of native plants for properly functioning rain garden, where the term ‘native’ referred to plants that are natural inhabitants of the particular region (Hitchmough, 2003). For example, Steiner and Domm (2012) recommended the use of North American native prairie species that evolved in seasonally moist prairies in rain gardens for their deep root systems that help them to survive summer dry spells and loose soil to improve infiltration. A series of rain garden guides such as the “Rain gardens: A how-to manual for homeowners (Bannerman & Considine, 2003) and “Rain gardens: a rain garden manual for South Carolina” (Giacalone, 2008) claimed ‘native only’ planting strategies that aimed to exclude the exotic species from rain garden components and to use only North American native plant species for their full integration into the local biotic community and better biodiversity restoration benefits over the introduced species. Native plants have gained popularity as most of the references argued that they have adapted to the indigenous growing conditions for thousands of years. As a result, they are resilient to a wider range of moisture levels and provide essential habitat for local wildlife that depend on these native plants. In contrast, exotic plant species are often claimed to be less suitable for the local environment and climate and may be highly invasive (Emanuel et al., 2010; Golon & Okay, 2014), which may result in intensive maintenance and resource consumption, and sometimes moral arguments because of their invasive behaviours that outcompete other species in rain garden components (Steiner & Domm, 2012).

However, Dunnett and Clayden (2007) pointed out that the native/non-native debate imposes a limiting factor on plant choices, partly because most references on plant options for rain gardens were North American sources thus only advise the use of American native species, and partly due to the biological fact that invasion related to certain plant traits which have high seed production or high dispersal ability other than their geographic origins. In fact, studies proved that some native species might generate biological invasion and result in decreased biodiversity (Reichard & White 2001; Sagoff, 2005), for instance, many invasions in the UK are not by exotic species, whilst some commonly seen but highly productive and ubiquitous native species such as *Pteridium aquilinum, Ulex, Rubus, Epilobium, Salix, Betula* and *Fraxinus* have been identified as damaging and causing habitat invasions in many areas in the UK (Pakeman & Marrs, 1993; Kendle & Rose, 2002).

Many cultivated exotic species neither were invasive in similar habitats outside of their inherent regions, nor reduced the local biodiversity (Owen, 1991; Kennedy & Southwood, 1984; Smith et al., 2005). Williams (1997), Lugo (1997), Hitchmough and Wagner (2011) argue that the non-natives provide ecosystem services in a similar way to native species in both natural habitats and urban ecosystems and have the possible functional benefits such as structural diversification, food supply and niche creation that ultimately benefit local biodiversity. These stated benefits derived from the use of introduced species may be particularly clear in altered urban environmental conditions, which are different from the predevelopment preferred by natives, such as urban developments and post-industrial soils (Kendle & Rose, 2002). For example, designed urban vegetation using introduced plant communities such as North American prairie and Sino-Himalayan *Primula* wet meadow were reported to support native European wildlife equally in urban areas to the native plant communities (Hitchmough & Dunnett, 2004; Hitchmough & Innes, 2007).

In fact, the responses of most species to environment out of their range are simply not known, where many experimental observations had refuted the arguments that natives grow better than non-natives in local environments (lines, 1987; Brown, 1997). For instance, the South African native *Gladiolus carneus* and *Gladiolus tristis* which grow in poorly drained areas in the wild are commonly cultivated in summer moist soils in the UK (Manning et al., 2002). Moreover, introduced plants are reportedly well adapted to the surroundings and tend to be tolerant of tough conditions (e.g. drought, inundation and poor soils) as long as the climate and conditions of a site match their cultural requirements (Dunnett & Clayden, 2007; Hansen & Stahl; 1993). However, the most ecologically similar sites may not be geographically closest, while species selected from a healthy population further afield may be as successful as the natives, or even perform better than the native species (Havens, 1998). For instance, mixed flower borders in the UK often use species from biogeographically similar regions, such as *Echinacea purpurea* and *Aster novi-angliae* from the North American native prairies and *Primula japonica* naturally occurs in Asian native wet meadows, and these species are found to withstand in variable weather conditions and soil conditions in the UK. Additionally, people may hesitate to use native species because they are colourless and dull, thus there is an increasing popularity in adapting plants imported from outside regions for a year-long appeal that bloom at various times of the year (particularly where the native flora may not be so grand or flower so late in the season). There are many plants from across the world that are suitable for use in rain gardens in different countries and are also visually dramatic in flowering display, for instance, the majority of plant species regularly recommended for North American rain garden applications are also highly desirable in European regions for their attractive displays (Dunnett & Clayden, 2007).

It is important to match plant species with the typical moisture conditions in rain gardens, however, only a handful of researchers made progress on plant selection for use in rain gardens based on experimental observations of plant growth in rain gardens. For instance, Vander Veen (2014) monitored the vegetative health of a series of trees, shrubs, forbs, ferns, grasses and vines in field retention facilities by visually judging the growth conditions of plants on saturated days and monitoring their available water to determine drought stress, as well as collecting the measured maximum number of consecutive days a plant species can tolerate saturated and dry soil; this data was then used to calibrate the feasibility of a list of preferred North American native plant species given by storm water management professionals that intended to use them in bioretention facilities. However, this study was not only limited by a lack of horticultural diagnostic analysis, but also based on separating the effects of flooding and drought on plants that tended to ignore the fact that the saturation and draining stages are constantly repeated to have the cyclic flooding cycle to affect planting establishment in typical rain garden conditions. Dylewski et al. (2011) repeated the saturation and draining cycle for pot plants to build the interval cyclic flooding situation to identify the growth and survival of three North-American native shrubs that were intended for use in rain gardens. However, the species studied were restricted to shrub species and lacked diversity. Due to the lack of studies on plant selections, designers can only have rather restricted plant lists with a small number of plants that can survive in most moisture conditions for choosing plants that they intend to use in rain gardens, whilst these lists often lack biodiversity.

3.1.3 Influence of waterlogging and drought on plants

Planting suggestions should be based on a proper understanding of plant responses and adaptations to the typical rain garden moisture dynamics. In fact, plant species have a remarkable diversity in tolerance to flooding and drought conditions. Therefore, plant selection for rain garden in which the habitat is similar to a transition zone between terrestrial system and wetland and adapting frequent switching between flooding, draining and drought is never a simple task. However, it is surprising that the current technical manuals and scientific research showed remarkably little evidence to fully reflect as to how cyclic flooding and potential prolonged drought in rain gardens may have influenced plant growth (herbaceous species in particular).

Flooding is abiotic stresses that may impose challenges to normal plant functioning (Jackson & Colmer, 2005), in which the frequency, duration and depth of inundation may alter the soil-plant relationships, thereby influencing productivity and species composition in vegetation (Blom & Voesenek, 1996; Casanova & Brock, 2000). Hypoxia and anoxia stress, as well as the possible high CO_2 concentration in root zone associated with waterlogging can severely damage the root metabolism and nutrient acquisition of plants, and thus inhibit the survival and growth of plant shoots and roots (Bailey-Serres & Colmer, 2014). Extremely inhibited diffusion of oxygen and carbon dioxide in saturated plant tissues would inhibit plant respiration and metabolic adaptations to cope with hypoxia and anoxia (Armstrong et al., 1994; Armstrong & Drew, 2002; Bailey-Serres & Colmer, 2014), and thus result in shortage of energy and carbohydrates in plants to hamper their productivities (Bailey-Serres & Voesenek, 2008). Excess water can also breakdown large soil aggregates into smaller particles, which may lead to a more compacted soil structure. Compacted soil is claimed to cause higher mechanical resistance to root and leave much less pore spaces leading to the reduction of oxygen availability, and thus stunt the plant growths (Engelaar, 1993; Kozlowski, 2012). Another constraint is that waterlogging could result in increased concentrations and rapid accumulation of toxic substances such as hormone ethylene in plant organs due to anaerobic metabolism by flooded tissues, which may lead to severe impacts on plant growth (Ponnamperuma, 1984; Mitchell & Rogers, 1985; Ernst, 1990; McKee & McKevlin, 1993; Voesenek & Sasidharan, 2013). Higher level of submergence reaching the shoot level or complete submergence can restrict plant photosynthesis by causing inadequate external carbon dioxide (CO_2) and shading (Jackson & Ram, 2003).

Plants have adaptive responses to waterlogging that enable survival in unique flooding regimes in their native habitats. Some plant species exploit life history adaptations to flooding, in which they conserve important life cycle events such as seed dispersal, germination, plant establishment and reproduction during flooding periods to avoid the negative impacts of submergence (Blom et al., 1990; Blom et al., 1994). For instance, some tolerant species may show low O_2 quiescence strategy that reducing the use of carbohydrates and energy or stop growth upon submergence to survive flooding events, and to rapidly complete their competitive growth between two flooding events (e.g. *Chenopodium rubrum*), while some species may postpone its flowering state outside of the flooding period (e.g. *Rumex palustris*) (Voesenek & Sasidharan, 2013). Some species exploit physiological adaptations to flooding such as fast shoot elongation that enable escape from submergence and to restore contact with the open air to thrive in flood-prone environments (Jackson, 1990; Van der Sman et al., 1991; Voesenek et al., 1992; Voesenek & Sasidharan, 2013). A few species may have aerenchyma that allows internal air pathways for the movement of oxygen and other gases in shoots and roots (Jackson & Colmer, 2005). Some amphibious plants (e.g. *Callitriche* and *Sparganium* species) have morphological features including thinner cuticles and leaf laminae, and flexible, thin and linear foliage, which may promote photosynthesis rates under waterlogged conditions and enable a faster plant uptake of the dissolved CO_2 linked to developmental plasticity (Mommer & Visser, 2005; Sand-Jensen et al., 1992).

Plants must maintain a favourable water status to sustain carbon assimilation vs. water conservation trade-off to avoid becoming carbon-limited and to sustain the assimilation rate (Cowan, 1977; Farquhar & Sharkey, 1982). Drought causing water deficit in soil, thereby becoming a major threat to plant health. Moreover, maintaining the water column in drought conditions may beget high stomatal conductance in plants and thus promotes high transpiration rate, which may not only increase the water loss through stomatal pores on the leaf surfaces, but also result in increasing tension in the water column and generate cavitation within the xylem to hamper plants' ability to transport water and nutrients (Skelton et al., 2015). All the stated drought effects may severely limit plant productivity and eventually lead to drought-induced plant mortality.

Certain plant species may develop functional traits to cope with drought threats. For instance, species with root system architecture of long root length and considerable root length density may help save water in deep soil profile and also improve root acquisition of water at depths in soil with available water to sustain plant survival and yields in extended drought periods (Comas et al., 2013). Plants with larger canopy size may have a greater transpirational water loss through their bigger leaf areas and are thus prone to drought-induced stress compare to smaller plants. However, some plant species may subside their stoma, or have foliage coated with wax or oil, which are beneficial to reduce transpiration to prolong survival in drought (Sangster & Parry, 1971; Yong-Rim et al., 2012). It is also worth noting that C_4 species (mainly grasses) have increased CO_2 assimilation and water use efficiency compare to C_3 species (to which the majority of plants belong, e.g. perennial forbs) (Lara & Andreo, 2011). Therefore, C_4 plants tend to have greater photosynthetic efficiency and productivity compared with C_3 plants in hot, dry environment and severe droughts (Edwards & Walker, 1983; Wards et al., 1999).

3.1.4 Recent methods in determining plant options in typical rain gardens

Plant selection for rain gardens should be based on the performances (i.e. growth and stress tolerance) of different plant types and specific species under the influence of the dynamic spatiotemporal moisture distribution in rain garden (i.e. cyclic flooding, potential drought and the gradient of moisture level through the margin-slope-bottom spatial structure). To choose appropriate plants for a rain garden, a test is needed to select candidate species in simulated cyclic flooding and drought conditions. Dylewski et al. (2011) soaked pot plants in a water bath for a certain period of time and took them out to allow sufficient draining, and the soaking and draining phases were repeated to build the interval cyclic flooding situation to determine the growth and survival of three North-American native shrubs (*Ilex glabra* ‘Shamrock’, *Itea virginica* 'Henry's Garnet' and *Viburnum nudum* 'Winterthur') that were intended for use in rain gardens. Dylewski et al. (2011) flooded pot plants for 0 day (non-flooded with regular irrigation to sustain the substrate per cent moisture at 25%), 3 days and 7 days, and were then taken out to drain for one week (without irrigation) until the next flood cycle began. It is noticeable that 7-day flood treatment was much longer than the specific flooding period that is allowed in typical rain gardens (4 days). In this study, shoot dry weight (SDW), root dry weight (RDW) and growth index (GI, i.e. [(height + widest width + width perpendicular to widest width) ÷3]) were used as indicator factors for determining the effect of cyclic flooding on the growth of three shrub species, and the survival rates of different species were also measured (Dylewski et al., 2011). Dylewski et al. (2011) found elevated mortality rates in all three shrub species and RDW, SDW and GI in all three species were significantly reduced because of cyclic flooding treatments, where the RDW, SDW and GI in plants from 3-day flood treatments were not statistically from plants from 7-day flood treatments. Dylewski et al. (2011) concluded the results were acceptable and all species were claimed to be tolerant of cyclic flooding.

However, the techniques involved in Dylewski’s work were unable to detect the stress a plant suffered in responding to cyclic flooding or drought. The traditional methods of plant stress detection include the measurements of the fresh weight/dry weight ratio in leaves, starch content, micro and macro nutrients, concentrations of amino acids, leaf and root electrolyte leakage and respiratory rate, etc. (Percival & Dixon, 1997; Figueiredo, 1985). However, many of these techniques have rather slow processes, and could be expensive and destructive, as well as being consistently augmented for their limited reliability and accuracy (Percival & Dixon, 1997; Figueiredo, 1985). Therefore, to understand the plant performance in typical rain garden conditions, a rapid, non-destructive and cost-efficient method is needed to detect stress in plants throughout their growth season under the effects of cyclic flooding and potential drought.

Furthermore, waterlogging and drought stresses either directly or indirectly decrease the photosystem activity and leaf photosynthetic efficiency prior to visible deteriorations in plants (Percival & Dixon, 1997). Soil waterlogging and submergence of plant tissues can inhibit photosynthesis (Jackson & Ram, 2003), and cause photoinhibition (i.e. the light-induced reduction in the photosynthetic capacity of a plant) (Percival & Dixon, 1997, Loll, 2005; Umena, 2011). A portion of the light energy absorbed by chlorophyll molecules in plant leaves drives photosynthesis, where the remaining excess energy can be re-emitted as light which refers to the leaf chlorophyll fluorescence (Maxwell & Johnson, 2000). The yield of chlorophyll fluorescence has been popularly used as an indicator of photosynthetic energy conversion in higher plants, where most investigations into the photosynthetic performance of plants under field conditions were completed with the use of chlorophyll fluorescence yield (Maxwell & Johnson, 2000). Photoinhibition can be detected from the reduction in the yield of chlorophyll fluorescence, so that it is used as an effective indicator of plant stress (Kooten & Snel, 1990). F_v/F_m ratio is one of the most used chlorophyll fluorescence measuring parameters (Maxwell & Johnson, 2000), where F_v refers to the difference between the measurements of the maximum level and minimum level of fluorescence yield, and F_m refers to the maximum level of fluorescence yield (Chowdhury et al., 2009; Kitajima & Butler, 1975; Ottander et al., 1995). The fluorescence yield reaches the maximum level in the absence of photochemical quenching, so that F_m is required to be measured in a dark-adapted leaf (Maxwell & Johnson, 2000). In recent years, leaf chlorophyll fluorescence measurements can be made using modulated plant efficiency analyser with the leaf poised in a known dark-adapted state (Rosyara et al., 2010), where the readings of the F_v/F_m ratio can be simply obtained from the analyser. A few studies adopted chlorophyll fluorescence as an effective indicator to evaluate waterlogging/drought stress in amenity plants (Pessarakli, 2005; Smethurst & Shabala, 2002; Smethurst et al., 2005). However, the use of leaf chlorophyll fluorescence for evaluating waterlogging/drought tolerance in the candidate plants under the stress of typical cyclic flooding and potential drought in rain garden remains unreported.

3.1.5 Research objectives

There have been neither replicated experiments that tested the feasibility of the recommended plant species for the saturation zones throughout the ‘dry-wet’ moisture gradient under the effects of typical cyclic flooding and potential drought in rain garden, nor data-based studies to support the much higher preference of native species over exotic species in the design guidance. Therefore, this study focuses on quantitatively understanding the effects of cyclic flooding and potential drought on different plant types (classified via assumed moisture sensitivities) and representative native/non-native species. There are four specific research objectives of carrying out this experiment:

1. Quantify the effects of cyclic flooding and drought on the survival, growth and stress of a range of representative perennial forbs and grasses (15 candidate species in total) intended for use in rain gardens.

2. Evaluate the suitability of a selected range of potential plant species for rain garden applications. Plants must be able to withstand periodic inundation with water, but also grow in normal conditions during dry periods.

3. Test the specific hypothesis proposed in most rain garden technical manuals that appropriate plants for different saturation zones (i.e. the margin, slope and basin bottom) and cyclic flooding or potential drought conditions in typical rain garden

depression structure is based on their moisture sensitivities. The test is based on the representative species' hydrologic responses to cyclic flooding and drought.

4. Test the feasibility of using exotic species (North American species and Asian species in particular) to expand the plant options in rain gardens.

3.2 Methods

This study followed the pot-in-pot methodology of Dylewski et al. (2011) that used periodic water bath and draining to simulate the cyclic flooding in typical rain gardens, as well as withheld irrigation to simulate drought, to test their effects on the survival and growth of pot plants of 15 candidate herbaceous species (11 Forbs and 4 grasses), as well as to detect stress in these selected plants by evaluating the measurements of chlorophyll fluorescence.

3.2.1 Site and materials

The study was conducted in an unheated, ventilated greenhouse situated at Norton Nursery (53°20'00.6"N, 1°27'44.9"W), Sheffield, UK. Over the course of the experiment a minimum temperature of 7.6 ℃ was recorded and a maximum air temperature of 34.3 ℃, while the daily relative humidity varied between 15.0 % and 89.8%. On 15 April 2013, Plants in 9cm pots of 15 perennial species were planted into 2-L freely drained pots with drainage holes. The plants were obtained from

Orchard Dene Nurseries, Lower Assendon, Henley-on-Thames, Oxfordshire, UK. A single plant of each species was planted in one pot, and there were 20 pots for each species. The plants were then watered every other day to maintain substrate moisture for a month to establish prior to treatments.

The artificial substrate that was used for this study was a mix of sharp sand (grain sizes range 0.06 to 4.0 mm, obtained from B&Q, Eastleigh, UK), seed bank free topsoil (screened to remove stones and big aggregates, obtained from Manor Cottages, Sheffield, UK) and sterilised peat (particle sizes range 0 to 22 mm, composted at Manor Cottages, Sheffield, UK) at a volume ratio of 5:2:3. Substrate sample taken in situ was sent to the Department of Geography, University of Sheffield on 30 July 2013 for analytical analysis to obtain the physical characteristics (e.g. percentage content of sand, silt and clay in the substrate, and porosity of the substrate) and chemical characteristics (e.g. percentage content of organic matter and pH). Double ring infiltration test was performed on 10 May 2013 to investigate infiltration behaviour of the used substrate following the experimental procedure provided by Lai and Ren (2007). In the present study, double ring infiltration test was conducted in a free-draining container with a surface area of 2000 mm × 1000 mm and water outlets on both sides. Substrate was filled to 300 mm depth in the container. The container was placed on flat ground and the surface area of the substrate was made flat. Two heavy metal infiltrometer rings (cylinders 200 mm high and of different internal diameters) were driven 50 mm into the substrate, where the sides of the infiltrometer rings were kept vertical. The smaller ring with an internal diameter of 100 mm was installed centred inside the outer ring with an internal diameter of 220 mm. Water was initially added to 50 mm depth into both rings at the same time. A Mariotte bottle (i.e. a device that automatically delivers a constant rate of flow from closed bottles) that had a height at 360 mm with a 200 mm inner diameter was used for maintaining the water level in the inner ring. The water level in the outer ring was adjusted manually so that it would match the water level in the inner ring. The Mariotte bottle that added water into the inner ring was graduated from 0 to 230 mm in 1 mm subdivisions, allowing visual readings. The rate of fall of the water depth in the Mariotte bottle was measured every minute. The measurement was stopped only when the rate of fall of the water depth per minute in the Mariotte bottle reached a constant value. The infiltration rate was then determined as the amount of water per surface area of the inner ring and time unit that penetrates the soil. Devices used for substrate infiltration rate measuring were provided from the Department of Geography, University of Sheffield and were shown in Fig. 3.2. Results showed that the substrate was gritty sandy loam (67.2% sand, 13.7% silt and 0.01% clay) with an organic matter content of 8.21% and a pH of 7.9. The substrate was free-draining with a porosity of 66.5% and a drainage rate of 5.7 cm/hour, which was ideal to the water infiltration and planting establishment in typical rain gardens (it was recommended from 0.25 cm/hour in clay loam soil to 21 cm/hour in sandy loam soil, Woelfle-Erskine & Uncapher, 2012).

Fig. 3.2. Devices used for the measurement of soil infiltration rate. Photo was taken by the author in May 2013. (Mariotte bottle was the one in the right of this photo, the other two bottles on the left in this photo were used for manually adding water into the outer ring.)

Following the instructions from most technical manuals and related literatures, most of the given species in this study were assumed to withstand the periodic/ seasonal inundation (seven species in total) or to withstand infrequent inundation (six species). Only the European native *Caltha palustris* was indigenous in regular saturated conditions, and the North American imported *Gaura lindheimeri* was assumed to be intolerant of inundation. These two species were chosen to represent the potential extremes of condition in a rain garden context. The 15 candidate herbaceous perennials and grasses in this study included five European native species, four species imported from North America and six species that are native to Asia (Table 3.1). The European native species and North American species were selected because they were either regularly used in rain gardens or proposed as being suitable in typical rain garden guides. The North American native species are selected particularly for their attractive late blooming flowers to support the flora’s landscape impact from late summer to autumn. Most of the selected Asian species naturally occur in Asian temperate wet meadow habitats. They were identified as being capable to adapt to moister soils which may become saturated during wetter periods but also may acclimate to dryer periods according to botanic documents (Brickell, 2008; Dunnett & Clayden, 2007; Hansen & Stahl, 1993; Hubbard, 1984; Steiner & Domm, 2012; Thomas, 1976), and were selected for their distinct visual appeal from native European meadows and prairie plants with as much emphasis on foliage textures as there was on flower colours.

Table 3.1. Selected species and their typical regions of origin and assumed moisture sensitivities (Brickell, 2008; Dunnett & Clayden, 2007; Hansen & Stahl, 1993; Hubbard, 1984; Steiner & Domm, 2012; Thomas, 1976)

Species		Typical region of origin	Assumed moisture sensitivity
Amsonia tabernaemontana var. salicifolia	Forb	E.USA	Infrequent inundation
Astilbe 'Purple Lance'	Forb	China	Periodic or seasonal inundation
Calamagrostis brachytricha	Grass	E. Asia	Periodic or seasonal inundation
Caltha palustris	Forb	Europe	Continuous inundation
Deschampsia flexuosa	Grass	Europe, Asia & N.E.USA	Periodic or seasonal inundation
Filipendula purpurea	Forb	E. Asia	Periodic or seasonal inundation
Gaura lindheimeri	Forb	N. America	Intolerant of inundation
Hemerocallis 'Golden Chimes'	Forb	E. Asia	Infrequent inundation
Iris sibirica	Forb	European	Infrequent inundation
Miscanthus sinensis	Grass	S.E. Asia	Periodic or seasonal inundation
Molinia caerulea	Grass	Europe	Periodic or seasonal inundation
Rudbeckia fulgida var. deamii	Forb	N. America	Infrequent inundation
Sanguisorba tenuifolia ‘Purpurea’	Forb	Asia	Infrequent inundation
Thalictrum aquilegifolium	Forb	Europe & Asia	Infrequent inundation
Veronicastrum virginicum	Forb	N. America	Periodic or seasonal inundation

Typical characteristics of each species used in this study and their preferred hydrological regimes are listed below, following the guide of a series of botanic documents and rain garden specific manuals (Brickwell, 2008; Dunnett & Clayden, 2007; Hansen & Stahl, 1993; Hubbard, 1984; Steiner & Domm, 2012; Thomas, 1976).

Amsonia tabernaemontana var. salicifolia A variant of *Amsonia tabernaemontana* which hails from Eastern USA. These perennials form tight clumps with upright bushy stems and narrow, ovate to elliptic, dark green leaves. *Amsonia tabernaemontana var. salicifolia* is easy to grow and its pale blue star-shaped flowers are loosely held on top of tall stems with a relatively long flowering period from spring to midsummer. It prefers moderately moist soils but copes well with drought, and grows in full sun or semi-shade.

Astilbe 'Purple Lance' A tall, vigorous *Astilbe chinensis* hybrid, which originates from China. It has handsome elliptic-ovate, 3-ternate leaves and slender purple-red panicles over mid-green leaves, flowering in late summer and early autumn. They usually grow in damp soil or are used for waterside plantings, and thrive in sun or shade.

Calamagrostis brachytricha A hardy perennial grass that is native to Eastern Asia. This compact clump-forming grass produces gentle arching green foliage, and has fluffy silvery-white flower heads that emerge in summer. The feathery flowers take on buttery shades in autumn and last well into winter. It grows best in moist to damp and fertile soil. It can thrive in full sun to shade growing conditions.

Caltha palustris A British and European native, marginal aquatic perennial with decumbent rhizomes that produce kidney-shaped and toothed, dark green leaves. It produces waxy yellow flowers in spring. It can withstand full submergence for short periods, but prefers shallow water and bog conditions.

Deschampsia flexuosa Tufted, often rhizomatous, evergreen perennial grass with thread-like, blue-green leaves, which is widely available in Europe, Asia and north-eastern USA. It has silver-tinted, purple or brown spikelets and open panicles in early and midsummer. It is naturally found in damp meadows and moorland, but can withstand dry to damp soil in sun or partial shade.

Filipendula purpurea Clump-forming perennial that is native to Eastern Asia. It bears pinnate, toothed leaves, with regularly 5- to 7-lobed, rounded to obovate terminal leaflets. It produces branching, crimson-purple stems bearing dense corymbs of pinky flowers in mid and late summer. It is fully hardy in damp to moist sites.

Gaura lindheimeri Bushy, clump-forming perennial that is native to North America. It produces lance-shaped and toothed leaves. It produces loose panicles of white to pink flowers from late spring to early autumn. It typically occurs naturally in dry, stony steppe and grows best in well-drained dry soil in full sun or partial shade.

Hemerocallis 'Golden Chimes' A free-flowering, evergreen perennial that is native to Eastern Asia. It bears narrow leaves and slender, well-branched and reddish brown scapes. It produces deep yellow flowers with reddish brown backs to outer tepals in early summer. It prefers fertile, moist but well-drained soil in full sun or semi-shade. Flowers will be reduced due to dry conditions.

Iris sibirica A European native, rhizomatous, beardless iris with narrow, grass-like leaves. It flowers in late spring and early summer with each branched stem bearing up to five flowers well above the foliage. All petals are blue-violet, while the background colour changes to white near the hafts. It prefers moist but well-drained soil in full sun or semi-shade.

Miscanthus sinensis A tall Southeastern Asian native deciduous, clump-forming, perennial grass with erect stems and erect or arching, linear, blue-green foliage. It produces silky-hairy spikelets, which are pale grey and tinted maroon or purple-brown in autumn. It naturally occurs in moist meadows and marshland, and is tolerant of a wide range of soil conditions in full sun.

Molinia caerulea A European native densely tufted perennial grass with clumps of flat, linear-oblong, mid-green leaves with purple bases. It bears dense, narrow panicles of purple spikelets on yellow-tinted stems from spring to autumn. It naturally inhabits damp moorland, but prefers to be cultivated in moist but well-drained soil in full sun or partial shade.

Rudbeckia fulgida var. deamii A variant of *Rudbeckia fulgida* that is native to North America. It is free-flowering with very hairy stems with long-pointed, oval-ovate, rough basal stem leaves. It needs to be grown in well-drained soil in full sun or partial shade.

***Sanguisorba tenuifolia* ‘Purpurea’** A cultivar of *Sanguisorba tenuifolia* that is native to Asia. It is spreading, clump-forming, rhizomatous perennial with upright or branched stems. It bears pinnate leaves, each composed of 13-21 narrowly oblong leaflets. It produces deep reddish to purplish flower spikes from midsummer to mid-autumn. It grows in fertile, well-drained, but not dry soil in full sun.

Thalictrum aquilegifolium A European and Asian native perennial with erect, clump-forming stems bearing 2- or 3-pinnate, hairless leaves composed of obovate, wavy-margined leaflets. It produces clustered, fluffy flowers with greenish white sepals, bright purple-pink or white stamens, and flat-topped, terminal panicles on glaucous stems in early summer. It grows in moist, humus-rich soil in partial shade.

Veronicastrum virginicum A North American native perennial with erect, hairless and unbranched stems bearing lance-shaped, pointed, toothed dark green leaves. It produces tubular, bluish purple flowers with protruding stamens, in slender, dense, terminal and axillary racemes. It grows in moderately fertile, moist soil in full sun or semi-shade.

3.2.2 Experimental design

Five single pot replicates of each of the species were given each of the experimental treatments. The cyclic flooding and drought treatments commenced on 28 May 2013. Treatments consisted of:

1. A non-flooded control group in which plants were carefully irrigated to maintain their substrate moisture from 0.20 $m^3 \cdot m^{-3}$ up to 0.25 $m^3 \cdot m^{-3}$ following the instructions from the work of Bailey (2009) and Dylewski et al. (2011) to keep the plants well watered in a mesic to moderately moist substrate, which was expressed in volumetric terms ($m^3 \cdot m^{-3}$, i.e. the absolute value of soil water content). Substrate moisture of topsoil (to 50 mm depth) per pot in this group was measured daily between 9 am and 10 am throughout the entire experiment using a handheld moisture probe (HH2 with SM200 moisture sensor, Delta-T Devices, Cambridge, United Kingdom). Soil moisture was measured for three measurements per substrate each time and calculated the mean value. Water was carefully added to maintain the substrate moisture at 0.20 to 0.25 $m^3 \cdot m^{-3}$ once the substrate dried to 0.15 $m^3 \cdot m^{-3}$,

2. A group that was flooded to substrate level for a one-day short interval cyclic flooding (1d group),

3. A group that was flooded to substrate level for a four-day longer-term interval cyclic flooding (4d group),

4. A dry group for which irrigation was withheld for the whole duration of the study (32 days). Substrate moisture of topsoil (to 50 mm depth) per pot in this group was measured daily between 9 am and 10 am throughout the entire experiment. Soil moisture was measured for three measurements per substrate each time and the mean value was calculated.

For the 1d and 4d flooded groups, plants were flooded to the level of the substrate by being placed in a saturated open top polyethylene water tanks with a surface area of 1000 mm by 500 mm (1.5 m^2) and a depth of 400 mm (Fig. 3.3) to simulate flooding conditions in a typical flooding cycle in rain gardens. These water tanks were ordered from LBS Horticulture Ltd., Colne, UK. In the 1d flooded group, plants were flooded to the level of the substrate for 1 day (24 hours), and were taken out for a 4-day draining after each inundation until the next flooding cycle was repeated. During the 4-day draining periods, plants from the 1d group were placed on flat concrete paving at 200 mm spacing, no irrigation was applied until the onset of the next flood cycle. In the 4d flooded group, plants were flooded to the substrate level for 4 days (96 hours), and were taken out for a 4-day draining after each inundation until the next flooding cycle was repeated. During the flooding treatments in the 4d flooded group, water was added into the water tanks every day to maintain the water table at the level of the substrate. During the 4-day draining periods, plants from the 4d group were placed on flat concrete paving at 200 mm spacing, no irrigation was applied until the next flood cycle began. The cyclic flooding treatments were concluded on 28 June 2013 (32 days in total). Plants in the 1d and 4d treatments experienced a total of seven and four flooding cycles, respectively.

Fig. 3.3. Plants in the water bath to simulate conditions in a typical flooding cycle. Photo was taken by the author in May 2013.

The control group with controlled irrigation and the drought treatments were also concluded on 28 June 2013 (32 days in total). All the plants from the control group and the dry group were placed on flat ground (bare soil) at 200 mm spacing. It is worth noting that the main focus of this study was to evaluate the growth rather than the further development of candidate species under the effect of a typical flooding cycle and potential drought in rain gardens. During the 32-day study, indoor temperature was fairly high at times, so that all the herbaceous plants tended to grow very fast to maximum (no visible growing tissues observed for at least one to two weeks).

Growth and survival of plants

At the termination of experiment, the roots and shoots of all the plants were harvested. Shoots in individual plants were removed from the root ball and dried at

80 ℃ for 48h to measure the shoot dry weight (SDW). Roots were washed free of substrate, and dried similarly to measure the root dry weight (RDW). Height and spread of each plant were recorded at experiment termination. In this study, plant height was determined from the bottom to the highest leaf apex. Each plant was observed from above to determine the plant length and width, and then the mean value was calculated to obtain spread. The survival rate of each species was also measured at experiment termination.

Stress detection via leaf chlorophyll fluorescence

In this study, leaf chlorophyll fluorescence was measured to evaluate stress in the selected plants. In this study, leaf chlorophyll fluorescence measurements referred to the F_v/F_m ratios. It was measured using a portable fluorescence spectrometer (Handy PEA, Hansatech Instruments, Norfolk, UK) (Fig. 3.4a), which was used in several previous studies (Chowdhury et al., 2009; Percival & Dixon, 1997). During the experiment, three leaves were randomly selected for measurements per plant to calculate the mean chlorophyll fluorescence value, and each leaf was tagged, ensuring that the measurements were taken from the same leaf for the whole duration of this study. To obtain the F_v/F_m, the selected leaves were dark-adapted for 30 min by attaching light exclusion leaf clips (Fig. 3.4b) to the leaf surface. These clips have a small shutter plate that was closed over the leaf to exclude light during the dark adaptive state. The shutter plates were opened after the 30 min dark adaptation, and the clips were attached with a sensor head (Fig. 3.4b) connected with the Handy PEA control unit to receive the chlorophyll fluorescence signal.

(a) **(b)**

Fig. 3.4. Handy PEA chlorophyll fluorescence monitoring instrument (a) and the leaf clips and sensor head (b). Photo was taken in May 2013

During this study, measurements of leaf chlorophyll fluorescence in each plant from the dry group were obtained at daily intervals (12 am). In the 1d and 4d flooded groups, leaf chlorophyll fluorescence was measured immediately after each flooding period and before the next flooding cycle started. Readings of leaf chlorophyll fluorescence in the control group were obtained at the same time when any of the other three groups were recorded.

In this study, the plant stress was estimated based on an optimal F_v/F_m value of 0.7. Lower F_v/F_m values than 0.7 indicated the onset of stress in selected plants and higher values than 0.7 indicated plants were healthy. Higher F_v/F_m values indicated better efficiency of photosynthesis and less plant stress (Maxwell & Johnson, 2000).

The use of 0.7 as the optimal F_v/F_m value was supported by numerous studies that chlorophyll fluorescence value <0.7 indicated the initiation of stress resulting in the effect of photoinhibition on photosynthesis, reduced growth and leaf necrosis (Close & Beadle, 2003; Curwiel & van Rensen, 1998; Delebecq et al., 2011; Demmig & Björkman, 1987; Morales et al., 1997; Tausz et al., 2004; Ribeiro et al., 2005; Weng et al. 2005). There are a few scientific documents that used an optimal F_v/F_m value higher than 0.7 for estimating plant stress, for instance, Maxwell and Johnson (2000) concluded that a F_v/F_m value ranging from 0.79 to 0.84 is the approximate optimal value, Endo et al. (1999) used 0.7-0.8 as the optimal F_v/F_m value to determine the recovery of photosynthesis from photodamage, Casanova-Katny et al. (2006) suggested 0.7 to 0.75 and Critchley (1998) concluded 0.725. However, it is clear that most documents claimed that chlorophyll fluorescence values <0.7 indicated the onset of stresses in plants.

Data analysis

Descriptive results for the survival and stress tolerances to cyclic flooding in each species are provided based on mortality rate at the termination of experiment and the time series chlorophyll fluorescence data (F_v/F_m). To analyse the effects of cyclic flooding on plant growth data (e.g. SDW, RDW, height and spread), one-way ANOVA is introduced. *F*-statistics and *P*-values obtained in one-way ANOVA are reported, while Welch corrections are used when Levene's test indicated non-homogeneity variance. Response variable residuals were explored for non-normality within dataset, but no clear evidence was found.

Mortality rates from the dry groups are reported. Instead of looking at the time series chlorophyll fluorescence (F_v/F_m), descriptive results of the number of days to onset of drought-induced stress (i.e. $F_v/F_m < 0.7$) and the soil moisture when stress was observed in each species are provided. Independent T-test is used for the comparison of plant growth data between the dry group and control group in each species. Because comparisons are not valid when different proportions of the population are alive or dead, growth data from plants that have died in dry group are excluded prior to the test. It is worth noting that dataset in this test was too small to look at normality. The previous analysis of cyclic flooding effects indicated no clear evidence of non-normality, therefore the normal distribution of residuals in this test was assumed. T-test is performed, when Levene's test indicated non-quality of variance, the adjusted *P*-value for the test is reported.

To further develop the understanding of the suitability and performance of the selected species, results of survival, physical growth and stress tolerance in each species are scored. Ranking methods for cyclic flooding and drought tolerance in individual species are presented in table 3.2. Friedman test is then used to see whether there were significant differences between species using all the ratings and produce a league table based on mean rank. Missing data was replaced by median value of the corresponding variable prior to Friedman test.

Table 3.2. Ranking method for cyclic flooding/drought tolerance in individual species

		Rank	Ranking method
	Survival	5	No mortality at termination of experiment.
		4	40% > Mortality rate ≥ 20% at termination of experiment.
		3	60% > Mortality rate ≥ 40% at termination of experiment.
		2	80% > Mortality rate ≥ 60% at termination of experiment.
		1	Mortality rate ≥ 80% at termination of experiment.
Cyclic flooding	**Growth data (e.g. SDW, RDW, Ht and Spd)[a]**	7	Corresponding data was significantly increased due to cyclic flooding treatment and main effect $P \leq 0.01$.
		6	Significant flooding-induced increase was found, while $0.01 < P \leq 0.05$.
		5	Possible increase, i.e. increase in corresponding data was found in cyclic flooding group, while $0.05 < P \leq 0.20$.
		4	Absolute non-significant effect of cyclic flooding treatments was determined, i.e. $0.20 \leq P$.
		3	Possible decrease, i.e. decrease in corresponding data was found in cyclic flooding group, while $0.05 < P \leq 0.20$.
		2	Significant flooding-induced decrease was found, while $0.01 < P \leq 0.05$.
		1	Corresponding data was significantly decreased due to cyclic flooding treatment and main effect $P \leq 0.01$.
	Chlorophyll fluorescence (F_v/F_m)	5	$F_v/F_m > 0.7$ was consistently found in1d and 4d groups throughout the experimental period.
		4	$F_v/F_m > 0.7$ was consistently found in 1d group throughout the experimental period. However, $F_v/F_m < 0.7$ can be occasionally found in 4d group, but overall performance was fairly satisfied.
		3	$F_v/F_m < 0.7$ can be occasionally found in both 1d and 4d groups, but overall performances were fairly satisfied.
		2	At least 50% of the total measurements from 4d group were < 0.7 throughout the experimental period. However, performance in 1d group was generally satisfied.
		1	At least 50% of the total measurements from both 1d and 4d groups were < 0.7 throughout the experimental period.
Drought	**Survival**	5	No mortality at termination of experiment.
		4	40% > Mortality rate ≥ 20% at termination of experiment.

	3	60% > Mortality rate ≥ 40% at termination of experiment.
	2	80% > Mortality rate ≥ 60% at termination of experiment.
	1	Mortality rate ≥ 80% at termination of experiment.
Growth data (e.g. SDW, RDW, Ht and Spd)[a]	4	No significant drought effect was determined on corresponding data. Main effect $P > 0.05$.
	3	Significant drought-induced decrease in corresponding data was found, while $0.05 \geq P > 0.01$.
	2	Significant drought-induced decrease in corresponding data was found, while $0.01 \geq P > 0.001$
	1	Significant drought-induced decrease in corresponding data was found, while $P \leq 0.001$
Number of days to the onset of plant stress (i.e. F_v/F_m < 0.7)	5	Mean value ≥ 9
	4	9 > mean value ≥ 7
	3	7 > mean value ≥ 5
	2	5 > mean value ≥ 3
	1	Mean value < 3
Soil moisture when F_v/F_m < 0.7	5	Mean value < 0.100
	4	0.200 > mean value ≥ 0.100
	3	0.300 > mean value ≥ 0.200
	2	0.400 > mean value ≥ 0.300
	1	Mean value ≥ 0.4

a: SDW=shoot dry weight, RDW=root dry weight, Ht=height, Spd=spread

Plant size (e.g. leaf area) is assumed to affect the adaptation of individual species during extended drought conditions. To prove this hypothesis, linear regression was carried out to identify the relationship between three canopy size factors (height, spread, and height × spread) and the mean number of days when onset of drought-induced stress ($F_v/F_m < 0.7$) was observed.

All statistical analyses in this study were performed with the SPSS 20.0 statistical package.

3.3 Results

3.3.1 Performances of selected species under cyclic flooding treatments

All the selected species had 100% survival rate in the control group and that of both 1d and 4d cyclic flooding treatments during the whole study. Mean values of shoot dry weight (SDW), root dry weight (RDW), mean height (Ht), and mean spread (Spd) in individual species, and the effects of duration of flooding cycle on these measurable parameters are demonstrated in Table 3.3. Overall, physical growths in 8 out of 15 selected species showed no significant changes in both 1d and 4d cyclic flooding groups compared to the regularly irrigated control group. These species included *Calamagrostis brachytrica, Caltha palustris, Deschampsia flexuosa, Filipendula purpurea, Hemerocallis 'Golden Chimes', Iris sibirica, Thalictrum aquilegifolium* and *Veronicastrum virginicum*. At least one of the measurable growth parameters in those of other selected species in this experiment was significantly affected by cyclic flooding treatments.

It appears that RDW (i.e. root growth) was the most sensitive growth parameter to the impacts of cyclic flooding, while the longer-term (4d) interval cyclic flooding significantly reduced the RDW in *Amsonia tabernaemontana var. salicifolia*, *Gaura lindheimeri,* and *Sanguisorba tenuifolia 'Purpurea'* (Table 3.3). However, it is worth noting that the mean RDW values in these three species obtained from the 1d group and the control group were not independent from each other (Table 3.3), which indicate the fair tolerance of their roots for the 1d cyclic flooding condition. It is also worth noting that significant increased RDW in *Molinia caerulea* was observed in 1d cyclic flooding treatment compare with the control group, while the mean RDW value obtained from the 4d group was not statistically different from those of 1d group and control group.

Canopy spread growth was also sensitive to cyclic flooding. Significant spread growth reduction in *Gaura lindheimeri* and *Rudbeckia fulgida var. deamii* was determined in the longer-term (4d) interval cyclic flooding group compared with the control group, while no statistical difference was found between the mean spread values of the short-term (1d) cyclic flooding group and the control group (Table 3.3). *Miscanthus sinensis* and *Molinia caerulea* had significantly decreased spread in the regularly irrigated control group compared with both 1d and 4d cyclic flooding groups, while no statistical difference was determined between the mean spread values of the 1d and 4d cyclic flooding groups (Table 3.3).

Astilbe 'Purple Lance' showed significantly decreased plant height in control group compared to both 1d and 4d cyclic flooding groups, which had no statistical differences between each other (Table 3.3). Similarly, *Miscanthus sinensis* had least height growth in the control group, while the 1d cyclic flooding treatment had the best plant height growth (Table 3.3). Mean height value of *Miscanthus sinensis* in 4d cyclic group was not statistically different from those of 1d group and control group (Table 3.3).

Only *Molinia caerulea* showed significantly decreased SDW in control group compared with that of 1d group, while the mean SDW value in 4d cyclic group was not statistically different from those of 1d group and control group (Table 3.3).

To sum up, *Astilbe 'Purple Lance', Miscanthus sinensis* and *Molinia caerulea* showed increases in plant growths due to the treatments of cyclic flooding. The other species affected by the treatments of flooding cycles (e.g. *Amsonia tabernaemontana var. salicifolia, Gaura lindheimeri, Rudbeckia fulgida var. deamii* and *Sanguisorba tenuifolia 'Purpurea'*) showed significant reduction in some of the growth parameters in the longer-term (4d) cyclic flooding group only, while fair tolerances to 1d cyclic flooding were indicated.

Table 3.3. Mean values of shoot dry weight (SDW), root dry weight (RDW), mean height (Ht), and mean spread (Spd) in each selected species, and the effect of duration of flooding cycle on these parameters. Plants in the 1d and 4d group experienced a total of seven and four flood cycles, respectively.

		Control	**Cyclic flooding**		***F*-statistic**	***P*-value**
			1d	**4d**		
Amsonia tabernaemontana var. salicifolia	**SDW (g)**	6.51a	6.86a	6.72a	0.096	0.909 (ns)
	RDW (g)	4.91b	5.85b	3.46a	6.444	0.013 (*)
	Ht (cm)	56.36a	60.08a	59.66a	0.578	0.576 (ns)
	Spd (cm)	24.05a	20.60a	21.26a	0.985	0.402 (ns)
Astilbe 'Purple Lance'	**SDW (g)**	26.72a	30.61a	28.11a	0.603	0.574 (ns)
	RDW (g)	48.87a	53.17a	56.28a	0.405	0.683 (ns)
	Ht (cm)	40.18a	55.06b	53.46b	5.715	0.018 (*)
	Spd (cm)	58.48a	57.32a	56.63a	0.133	0.877 (ns)
Calamagrostis brachytrica	**SDW (g)**	8.89a	10.05a	8.18a	1.763	0.213 (ns)
	RDW (g)	27.76ab	29.41b	21.11a	3.660	0.057 (ns)
	Ht (cm)	61.94a	56.36a	55.44a	3.065	0.115 (ns)
	Spd (cm)	58.55a	58.29a	51.55a	1.476	0.267 (ns)

Caltha palustris	**SDW (g)**	3.66a	3.15a	3.19a	1.080	0.370 (ns)
	RDW (g)	10.44a	9.75a	8.51a	0.687	0.522 (ns)
	Ht (cm)	14.20a	14.04a	15.18a	0.235	0.794 (ns)
	Spd (cm)	26.32a	24.51a	22.89a	1.579	0.246 (ns)
Deschampsia flexuosa	**SDW (g)**	12.34a	13.02a	14.12a	0.371	0.702 (ns)
	RDW (g)	3.24a	2.57a	3.71a	1.550	0.278 (ns)
	Ht (cm)	65.86a	64.78a	64.54a	0.722	0.506 (ns)
	Spd (cm)	64.41a	62.02a	63.40a	0.509	0.614 (ns)
Filipendula purpurea	**SDW (g)**	14.09a	15.18a	16.93a	4.130	0.065 (ns)
	RDW (g)	44.64a	53.49a	41.91a	1.667	0.230 (ns)
	Ht (cm)	37.32a	40.42ab	47.80b	3.094	0.082 (ns)
	Spd (cm)	39.34a	38.74a	41.41a	1.342	0.298 (ns)
Gaura lindheimeri	**SDW (g)**	11.43a	12.12a	12.13a	0.329	0.724 (ns)
	RDW (g)	8.72b	7.32ab	6.35a	4.439	0.036 (*)
	Ht (cm)	94.38a	81.08a	81.82a	1.267	0.317 (ns)
	Spd (cm)	29.41b	24.85ab	21.94a	6.934	0.010 (**)

***Hemerocallis* 'Golden Chimes'**	**SDW (g)**	14.91a	16.42a	16.01a	0.388	0.686 (ns)
	RDW (g)	19.96a	21.91a	16.64a	0.945	0.416 (ns)
	Ht (cm)	71.78a	70.16a	74.2a	0.338	0.720 (ns)
	Spd (cm)	50.64a	50.30a	46.78a	0.514	0.611 (ns)
Iris sibirica	**SDW (g)**	13.20a	16.60b	14.39ab	3.068	0.084 (ns)
	RDW (g)	26.64a	29.78a	24.30a	0.467	0.638 (ns)
	Ht (cm)	71.86a	74.78a	77.26a	1.539	0.254 (ns)
	Spd (cm)	38.55a	39.99a	42.34a	0.461	0.641 (ns)
Miscanthus sinensis	**SDW (g)**	14.00a	18.41a	18.05a	0.760	0.489 (ns)
	RDW (g)	34.44a	39.16a	32.59a	0.703	0.514 (ns)
	Ht (cm)	83.60a	100.36b	93.68ab	4.800	0.029(*)
	Spd (cm)	41.96a	82.18b	83.50b	29.658	<0.001 (***)
Molinia caerulea	**SDW (g)**	2.26a	4.54b	3.55ab	7.491	0.008 (**)
	RDW (g)	2.03a	4.19b	2.72ab	11.184	0.011 (*)
	Ht (cm)	36.92a	45.10a	42.16a	2.296	0.143 (ns)
	Spd (cm)	13.40a	29.70b	28.90b	31.742	<0.001 (***)

Rudbeckia fulgida var. deamii	**SDW (g)**	12.78a	11.66a	11.82a	0.657	0.536 (ns)
	RDW (g)	10.77a	9.94a	9.39a	1.097	0.365 (ns)
	Ht (cm)	34.44a	38.54a	34.84a	2.535	0.121 (ns)
	Spd (cm)	28.13b	25.33ab	23.82a	7.661	0.007 (**)
Sanguisorba tenuifolia 'Purpurea'	**SDW (g)**	9.83b	8.95a	8.30a	0.704	0.514 (ns)
	RDW (g)	10.84b	8.43a	7.93a	4.587	0.033 (*)
	Ht (cm)	35.70a	41.54a	34.68a	1.104	0.363 (ns)
	Spd (cm)	37.55a	35.37a	34.48a	1.509	0.260 (ns)
Thalictrum aquilegifolium	**SDW (g)**	3.60a	3.42a	3.03a	0.520	0.616 (ns)
	RDW (g)	5.17a	5.14a	3.17a	2.658	0.111 (ns)
	Ht (cm)	35.76a	37.82a	31.72a	0.376	0.699 (ns)
	Spd (cm)	25.35a	21.81a	23.84a	1.330	0.301 (ns)
Veronicastrum virginicum	**SDW (g)**	10.81a	11.00a	10.95a	0.026	0.975 (ns)
	RDW (g)	16.61a	18.27a	16.78a	0.468	0.637 (ns)
	Ht (cm)	103.48a	97.04a	83.42a	1.818	0.204 (ns)
	Spd (cm)	24.10a	25.33a	25.30a	0.101	0.905 (ns)

Lowercase letters denote mean separation within columns; means with the same letter do not differ significant from each other.

ns = not significant, *=between 0.05 and 0.01 **= between 0.01 and 0.001 and ***=<0.001

To understand the stress exhibited by the different species throughout the repeated flooding cycles and extended drought during the whole study, measurements of leaf chlorophyll fluorescence (F_v/F_m) are also observed individually by species and are showed in Fig. 3.5. Due to technical issues, it was not possible to obtain measurements of leaf chlorophyll fluorescence in *Deschampsia flexuosa*. Interpretations of what the results mean were made for each species.

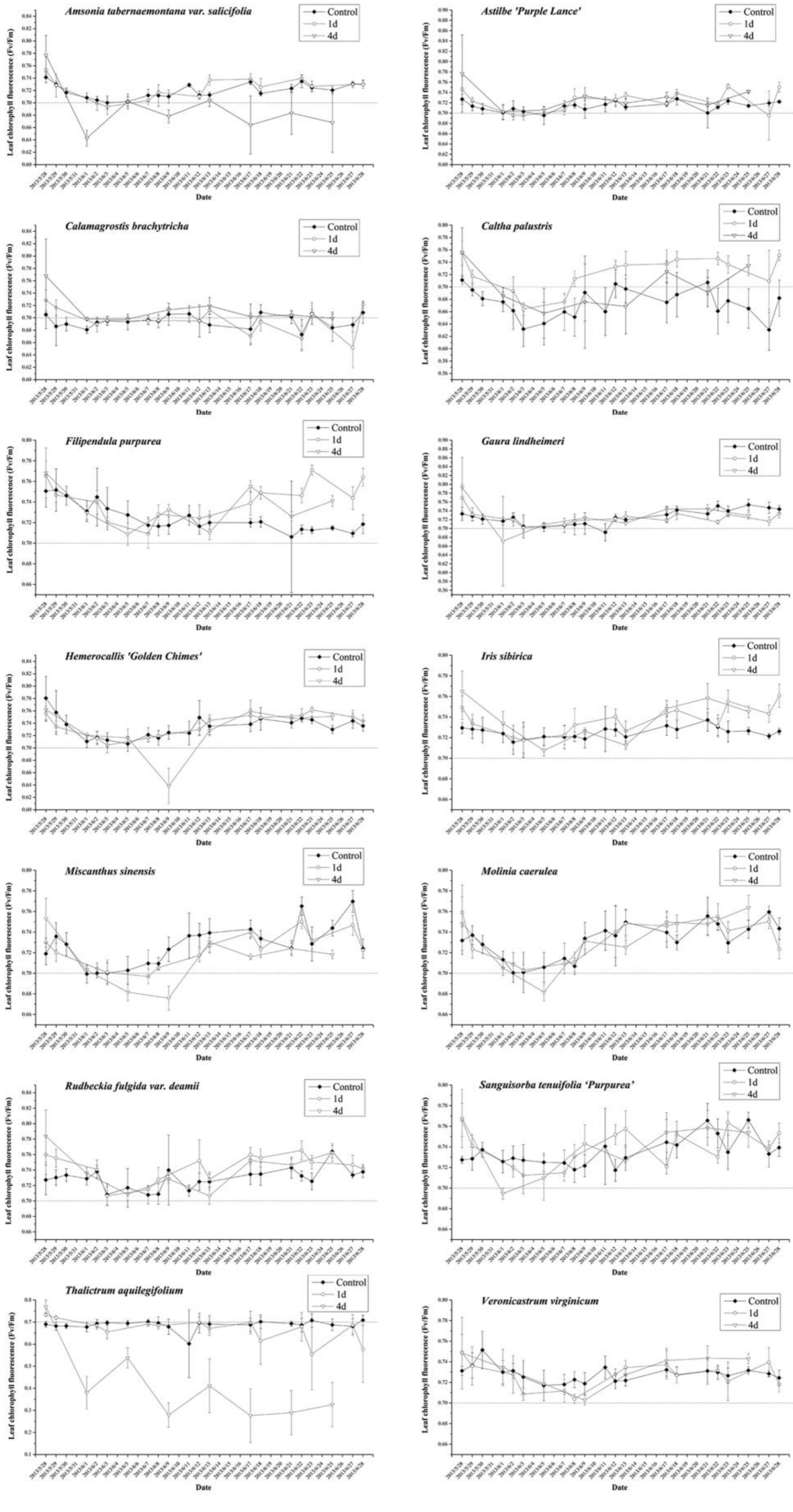


Fig. 3.5. Changes in the mean values of chlorophyll fluorescence from the control, 1d and 4d treatments in individual species.

Error bars represent standard error.

Filipendula purpurea, Iris sibirica, Rudbeckia fulgida var. deamii and *Veronicastrum virginicum* showed the best stress tolerance to cyclic flooding among all selected species. The time series F_v/F_m from all groups (e.g. control, 1d and 4d cyclic flooding) in the four species was consistently above 0.7 throughout the whole study (Fig. 3.5). Such results generally matched with their physical growths that no significant effects of cyclic flooding were indicated, except that decrease of canopy spread in *Rudbeckia fulgida var. deamii* was showed in only 4d cyclic flooding treatment. The general trend that recovery of photosynthesis efficiency (F_v/F_m) in *Filipendula purpurea* and *Iris sibirica* was found during flooding periods, whereas F_v/F_m recovery in *Rudbeckia fulgida var. deamii* and *Veronicastrum virginicum* was observed during draining stages (Fig. 3.5).

F_v/F_m profile in *Gaura lindheimeri, Hemerocallis 'Golden Chimes', Molinia caerulea* and *Sanguisorba tenuifolia 'Purpurea'* only occasionally fell below 0.7 in 4d cyclic flooding group, but the overall performances were good (Fig. 3.5). The results showed these species had successfully established their tolerances to cyclic flooding stress. Results from *Hemerocallis 'Golden Chimes'*, *Molinia caerulea* and *Sanguisorba tenuifolia 'Purpurea'* generally matched with their physical growths. It is worth noting that three physical growth measurements including shoot/root dry weight and canopy spread in *Molinia caerulea* were found promoted by cyclic flooding. Considering the fact that this species showed increasing F_v/F_m during flooding stages (Fig. 3.5), *Molinia caerulea* should be one of the most robust species in cyclic flooding conditions among all selected species. *Sanguisorba tenuifolia ‘Purpurea’* also had increasing chlorophyll fluorescence during flooding stages, whereas recovering F_v/F_m during draining stages was observed in *Gaura lindheimeri* and *Hemerocallis 'Golden Chimes'* (Fig. 3.5). However, the fairly good cyclic flooding stress tolerance showed in *Gaura lindheimeri* did not match with its physical growth performance, in which significant reduction of root dry weight and canopy spread was determined due to the effects of longer-term (4d) cyclic flooding treatment.

Flooding-induced stress (i.e. $F_v/F_m < 0.7$) could be occasionally observed in both 1d and 4d cyclic flooding groups in *Astilbe 'Purple Lance'* and *Miscanthus sinensis* (Fig. 3.5), which showed fair stress tolerances in the two species. However, it appears that the observed stresses in the two species were found during the draining stages (Fig. 3.5). Furthermore, considering that fact that their height growths and the spread of *Miscanthus sinensis* were increased due to cyclic flooding treatments, the two species actually showed vigorousness under the effects of cyclic flooding. *Astilbe 'Purple Lance'* showed recovering F_v/F_m during flooding stages, while recovery was observed during draining periods in *Miscanthus sinensis* (Fig. 3.5).

At least 50% of the total F_v/F_m measurements from 4d group in *Amsonia tabernaemontana var. salicifolia* and *Caltha palustris* were less than 0.7, while more than half of the measurements from both 1d and 4d cyclic flooding group in *Calamagrostis brachytrica* and *Thalictrum aquilegifolium* were found below 0.7 (Fig.

3.5). Generally, these species showed poor stress tolerances to cyclic flooding treatments, and could only recover their photosynthesis efficiency during the draining stages. Such performances in *Amsonia tabernaemontana var. salicifolia* and *Thalictrum aquilegifolium* did not match with their physical growth results that most measurable parameters were not significantly affected by cyclic flooding. In addition, many leaves of *Thalictrum aquilegifolium* turned purple in the 4d group during the third flooded treatment, where obvious leaf necrosis were found on 17 June 2013 (end of the third flooding) (Fig. 3.6), which also indicated extreme plant stress and matched the rather low level of chlorophyll fluorescence in this species caused by the longer-term (4d) interval cyclic flooding. In contrast, *Calamagrostis brachytrica* and *Caltha palustris* showed stress due to the shortages of soil moisture during the draining stages, and only recovered chlorophyll fluorescence in waterlogged or damp soils (Fig. 3.5).

Fig. 3.6. Visible damages in *Thalictrum aquilegifolium* from the 4d group. Photo was taken by the author on 17 June 2013.

Performances of survival, physical growths and stress tolerance in each species are scored based on the observations stated previously, and results are presented in Table

3.4. Friedman test was applied on these ordinal-scale data, which indicates that significant differences between species using the ratings (*P* = 0.004). A league table base on mean rank is thus presented to show the level of suitability across different species in cyclic treatments (Table 3.5).

Table 3.4. Ranked cyclic flooding performances in individual species, including survival, shoot dry weight (SDW), root dry weight (RDW), height (Ht), spread (Spd) and stress tolerance

Species	Survival	SDW	RDW	Ht	Spd	Stress tolerance
Amsonia tabernaemontana var. salicifolia	5	4	2	4	4	2
Astilbe 'Purple Lance'	5	4	4	6	4	3
Calamagrostis brachytrica	5	4	3	3	4	1
Caltha palustris	5	4	4	4	4	2
Deschampsia flexuosa	5	4	4	4	4	/*
Filipendula purpurea	5	5	4	5	4	5
Gaura lindheimeri	5	4	2	4	1	4
Hemerocallis 'Golden Chimes'	5	4	4	4	4	4
Iris sibirica	5	5	4	4	4	5
Miscanthus sinensis	5	4	4	6	7	3
Molinia caerulea	5	7	6	5	7	4
Rudbeckia fulgida var. deamii	5	4	4	5	1	5
Sanguisorba tenuifolia 'Purpurea'	5	4	2	4	4	4
Thalictrum aquilegifolium	5	4	3	4	4	1
Veronicastrum virginicum	5	4	4	4	4	5

*: F_v/F_m was not obtained in *Deschampsia flexuosa* due to technical issues.

Table 3.5. League table based on mean rank of individual species' performance in cyclic flooding treatments. Higher-scored species showed better suitability and overall performance.

Species	Mean Rank
Calamagrostis brachytrica	4.92
Gaura lindheimeri	5.5
Amsonia tabernaemontana var. salicifolia	5.67
Thalictrum aquilegifolium	5.75
Sanguisorba tenuifolia 'Purpurea'	6.58
Caltha palustris	7
Deschampsia flexuosa	7.92
Hemerocallis 'Golden Chimes'	7.92
Rudbeckia fulgida var. deamii	8.58
Veronicastrum virginicum	8.67
Astilbe 'Purple Lance'	8.75
Iris sibirica	9.83
Miscanthus sinensis	9.83
Filipendula purpurea	10.83
Molinia caerulea	12.25

3.3.2 Performances of selected species under drought treatments

Mortality in different species from the dry group was presented in Table 3.6. It could be seen that *Hemerocallis 'Golden Chimes'* and *Iris sibirica* had the best survival (no mortality) among all the species, whereas no specimen in *Caltha palustris* and *Filipendula purpurea* was alive at the termination of the experiment. Mortality rates in the other species were varied, however, only *Amsonia tabernaemontana var. salicifolia* and *Deschampsia flexuosa* showed remarkable mortality rates that were greater than 50% (80% and 60%, respectively).

Table 3.6. Mortality rate in individual species from dry group

Species	Mortality rate (%)
Amsonia tabernaemontana var. salicifolia	80
Astilbe 'Purple Lance'	40
Calamagrostis brachytrica	40
Caltha palustris	100
Deschampsia flexuosa	60
Filipendula purpurea	100
Gaura lindheimeri	20
Hemerocallis 'Golden Chimes'	0
Iris sibirica	0
Miscanthus sinensis	40
Molinia caerulea	40
Rudbeckia fulgida var. deamii	40
Sanguisorba tenuifolia 'Purpurea'	20
Thalictrum aquilegifolium	20
Veronicastrum virginicum	40

Mean values of shoot dry weight (SDW), root dry weight (RDW), mean height (Ht), and mean spread (Spd) in individual species from dry group and control group were compared and results were presented in Table 3.7. As stated previously, only alive specimen's growth parameters were used for calculating means. Therefore, it is not possible to make comparison between dry group and the control group in *Caltha palustris* and *Filipendula purpurea*.

Table 3.7. Mean values of shoot dry weight (SDW), root dry weight (RDW), mean height (Ht), and mean spread (Spd) in each selected species, and the effect of drought on these parameters.

		Control	Dry	*t*-statistic	*P*-value
Amsonia tabernaemontana var. salicifolia	**SDW (g)**	6.51a	5.69a	0.675	0.537 (ns)
	RDW (g)	4.91a	5.14a	-0.260	0.808 (ns)
	Ht (cm)	4.92a	55.90a	0.054	0.96 (ns)
	Spd (cm)	4.93a	22.85a	0.204	0.848 (ns)
Astilbe 'Purple Lance'	**SDW (g)**	26.72b	17.17a	6.883	<0.001 (***)
	RDW (g)	48.87b	38.49a	2.556	0.043 (*)
	Ht (cm)	40.18a	41.43a	-0.446	0.672 (ns)
	Spd (cm)	58.48b	47.00a	7.662	<0.001 (***)
Calamagrostis brachytrica	**SDW (g)**	8.89a	7.74a	1.164	0.289 (ns)
	RDW (g)	27.76b	22.24a	3.359	0.015 (*)
	Ht (cm)	61.94b	53.53a	4.372	0.005 (**)
	Spd (cm)	58.55a	53.37a	1.138	0.298 (ns)
Deschampsia flexuosa	**SDW (g)**	12.34b	8.57a	2.739	0.041 (*)

	RDW (g)	3.24a	1.45a	1.325	0.242 (ns)
	Ht (cm)	65.86b	17.45a	27.725	<0.001 (***)
	Spd (cm)	64.41b	17.05a	19.373	<0.001 (***)
Gaura lindheimeri	**SDW (g)**	11.43a	9.00a	2.328	0.053 (ns)
	RDW (g)	8.72b	4.70a	3.898	0.006 (**)
	Ht (cm)	94.38b	58.60a	7.676	<0.001 (***)
	Spd (cm)	29.41b	21.89a	2.687	0.031 (*)
Hemerocallis 'Golden Chimes'	**SDW (g)**	14.91b	11.42a	2.800	0.023(*)
	RDW (g)	19.96a	16.31a	1.910	0.093 (ns)
	Ht (cm)	71.78b	45.98a	6.947	<0.001 (***)
	Spd (cm)	50.64b	40.69a	3.786	0.016 (*)
Iris sibirica	**SDW (g)**	13.20b	11.45a	2.429	0.041 (*)
	RDW (g)	26.64a	23.42a	0.714	0.511 (ns)
	Ht (cm)	71.86a	68.70a	1.280	0.236 (ns)
	Spd (cm)	38.55a	33.50a	2.070	0.089 (ns)

	SDW (g)	14.00a	8.70a	1.835	0.116 (ns)
	RDW (g)	34.44a	27.53a	1.839	0.116 (ns)
Miscanthus sinensis	**Ht (cm)**	83.50b	58.17a	17.655	<0.001 (***)
	Spd (cm)	41.69b	30.83a	3.980	0.007 (**)
	SDW (g)	2.26a	1.60a	2.194	0.071 (ns)
	RDW (g)	2.03a	1.50a	1.574	0.167 (ns)
Molinia caerulea	**Ht (cm)**	36.92b	17.86a	15.904	<0.001 (***)
	Spd (cm)	13.40a	14.42a	-0.840	0.447 (ns)
	SDW (g)	12.78b	8.90a	4.117	0.011 (*)
	RDW (g)	10.77a	9.50a	0.812	0.448 (ns)
Rudbeckia fulgida var. deamii	**Ht (cm)**	34.44b	23.43a	6.595	0.001 (**)
	Spd (cm)	28.13a	25.40a	2.234	0.067 (ns)
	SDW (g)	9.83a	7.95a	1.240	0.255 (ns)
	RDW (g)	10.84b	7.46a	2.469	0.043 (*)
Sanguisorba tenuifolia ‘Purpurea’	**Ht (cm)**	35.70b	20.13a	7.388	0.002 (**)
	Spd (cm)	37.55a	35.63a	1.403	0.23 (ns)

Thalictrum aquilegifolium	**SDW (g)**	3.60a	3.81a	-0.230	0.824 (ns)
	RDW (g)	5.18a	4.04a	1.222	0.283 (ns)
	Ht (cm)	36.92a	35.90a	0.465	0.656 (ns)
	Spd (cm)	24.77a	24.13a	0.348	0.738 (ns)
Veronicastrum virginicum	**SDW (g)**	10.81b	8.47a	3.200	0.019 (*)
	RDW (g)	16.61a	14.12a	1.659	0.148 (ns)
	Ht (cm)	103.48a	79.97a	2.001	0.092 (ns)
	Spd (cm)	24.10a	24.28a	-0.056	0.957 (ns)

Lowercase letters denote mean separation within columns; means with the same letter do not differ significant from each other.

ns = not significant, *=between 0.05 and 0.01 **= between 0.01 and 0.001 and ***=<0.001

Only *Amsonia tabernaemontana var. salicifolia* and *Thalictrum aquilegifolium* had no physical growth parameters affected by extended drought (Table 3.7). The rest of the selected species generally possessed intolerance of drought during this study in at least one of the physical growth parameters (i.e. SDW, RDW, height and spread) was significantly damaged by drought (Table 3.7). Shoot dry weight (SDW) in *Astilbe 'Purple Lance', Deschampsia flexuosa, Hemerocallis 'Golden Chimes', Iris sibirica, Rudbeckia fulgida var. deamii* and *Veronicastrum virginicum* showed significant decrease compared to the regularly irrigated control group (Table 3.7). Most species, excluding *Amsonia tabernaemontana var. salicifolia, Astilbe 'Purple Lance', Iris sibirica, Thalictrum aquilegifolium* and *Veronicastrum virginicum*, showed significant drought-induced reductions in plant heights (Table 3.7). *Astilbe 'Purple Lance'*, *Deschampsia flexuosa*, *Gaura lindheimeri*, *Hemerocallis 'Golden Chimes'* and *Miscanthus sinensis* showed significant decreased spread growths due to drought (Table 3.7). Significant drought-induced reductions of root dry weights were only observed in *Astilbe 'Purple Lance', Calamagrostis brachytrica, Gaura lindheimeri* and *Sanguisorba tenuifolia 'Purpurea'* (Table 3.7).

To understand the level of plant stress under drought conditions in each species, the number of days and the soil moisture when plant stress (i.e. $F_v/F_m < 0.7$) was observed in each species was obtained and showed in Fig. 3.7 and Fig. 3.8, respectively.

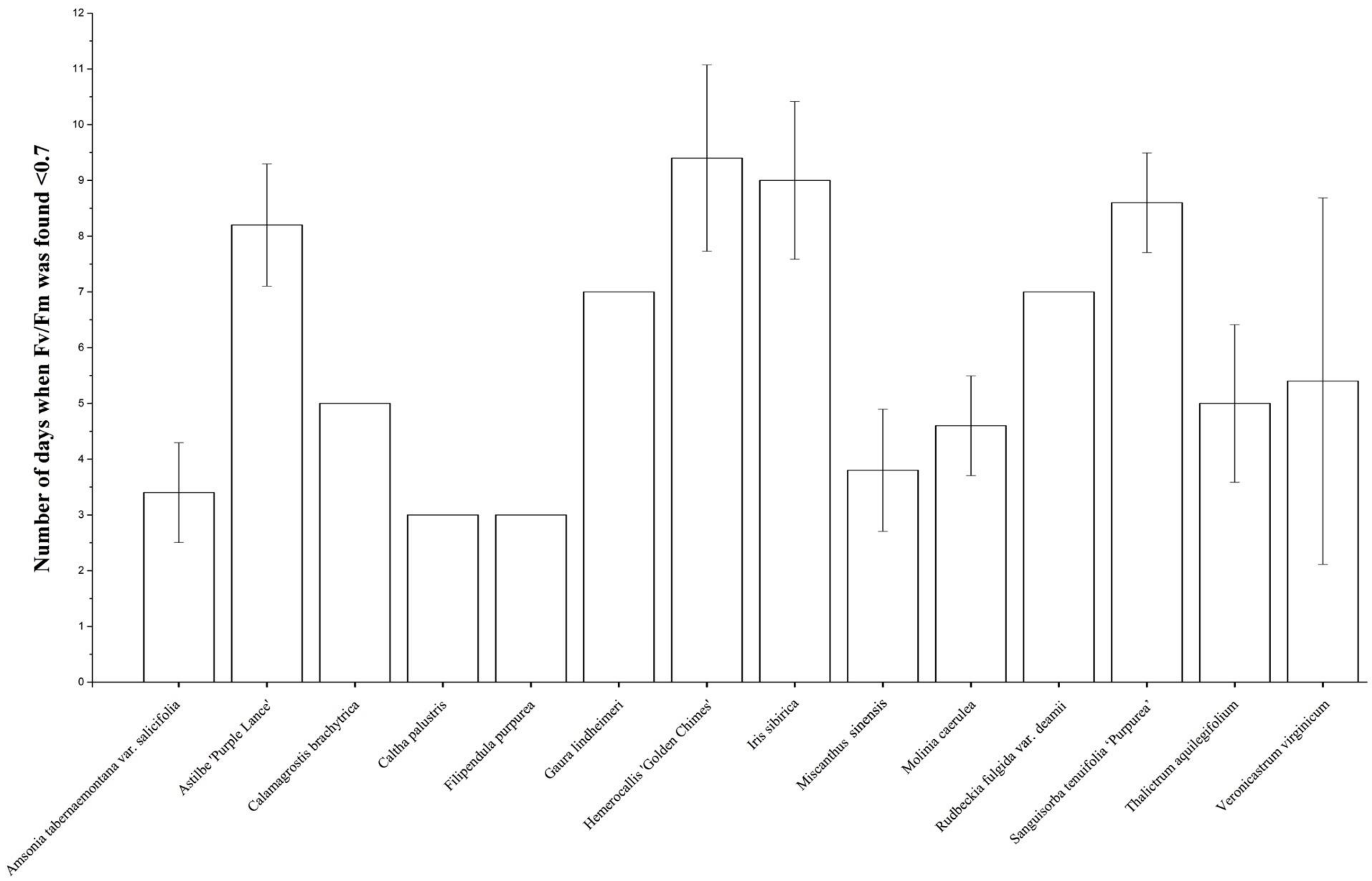


Fig. 3.7. Number of days when mean value of chlorophyll fluorescence was found below 0.7 in each species. Error bars represent standard error.

* Column without error bar means the initiation of stresses ($F_v/F_m < 0.7$) in the five plants in the species were detected on the same day.

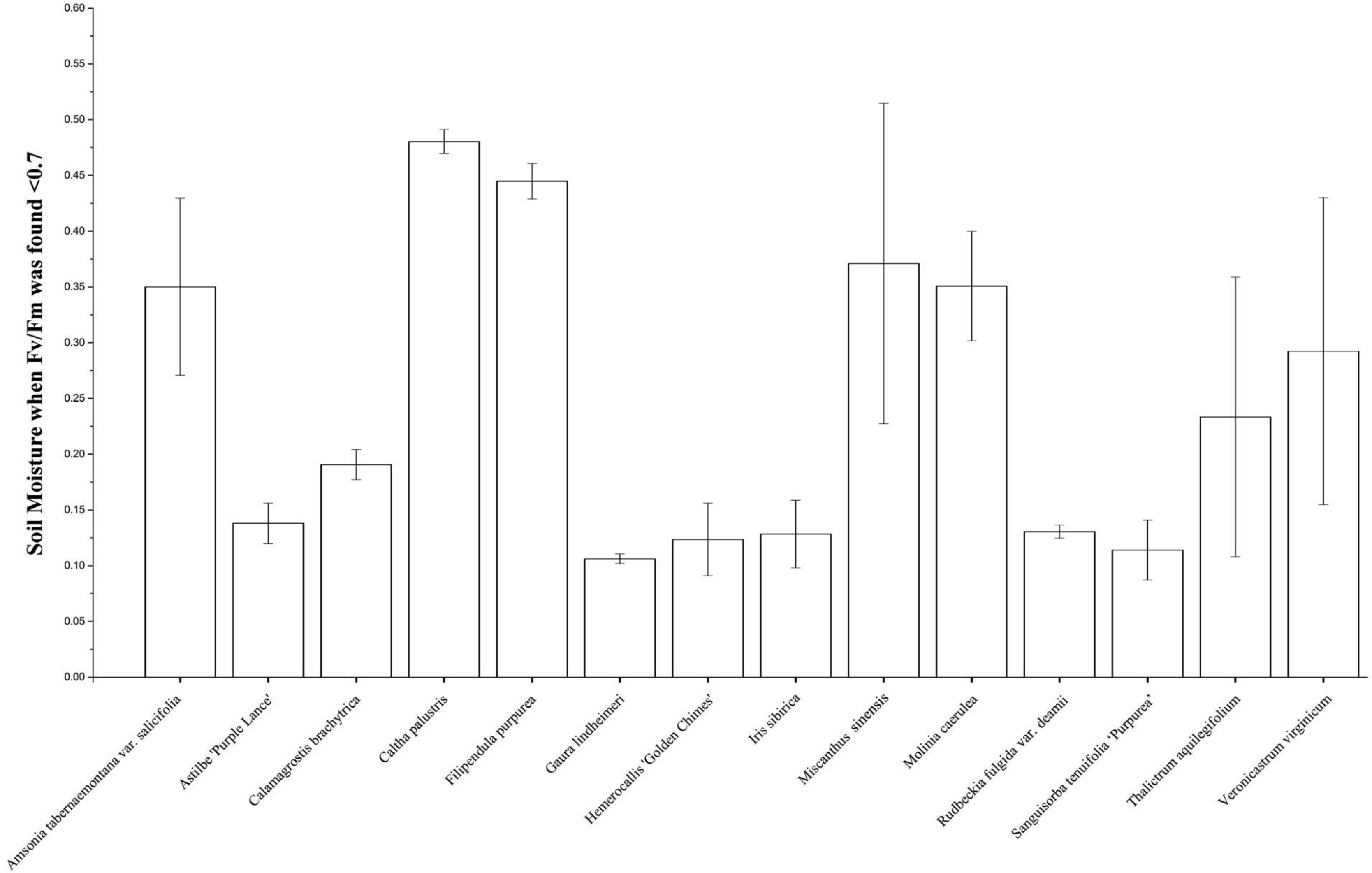


Fig. 3.8. Soil moisture when mean value of chlorophyll fluorescence was found below 0.7 in each species. Error bars represent standard error.

Fig. 3.7 shows that *Hemerocallis 'Golden Chimes'* and *Iris sibirica* had the longest tolerance duration (9.4 and 9 days, respectively) among all selected species until drought-induced stress (i.e. $F_v/F_m < 0.7$) was observed, followed by *Sanguisorba tenuifolia 'Purpurea'* (8.6 days), *Astilbe 'Purple Lance'* (8.2 days), *Gaura lindheimeri* and *Rudbeckia fulgida var. deamii* (7 days), *Veronicastrum virginicum* (5.4 days), *Calamagrostis brachytricha* and *Thalictrum aquilegifolium* (5 days), *Molinia caerulea* (4.6 days), *Miscanthus sinensis* (3.8 days), *Amsonia tabernaemontana var. salicifolia* (3.4 days), and *Caltha palustris and Filipendula purpurea* (3 days).

According to Fig. 3.8, the highest mean value of soil moisture for the onset of drought-induced stress (i.e. $F_v/F_m < 0.7$) was observed in *Caltha palustris* (0.480), followed by *Filipendula purpurea* (0.445), *Miscanthus sinensis* (0.371), *Molinia caerulea* (0.351), *Amsonia tabernaemontana var. salicifolia* (0.350), *Veronicastrum virginicum* (0.292), *Thalictrum aquilegifolium* (0.233), *Calamagrostis brachytricha* (0.191), *Astilbe 'Purple Lance'* (0.138), *Rudbeckia fulgida var. deamii* (0.131), *Iris sibirica* (0.128), *Hemerocallis 'Golden Chimes'* (0.124), *Sanguisorba tenuifolia 'Purpurea'* (0.114) and *Gaura lindheimeri* (0.106).

The relationship between the plant size (e.g. height, spread and height × spread) and the number of days to the onset of plant stress (i.e. $F_v/F_m < 0.7$) is presented in Fig 3.9 and Fig. 3.10. There was no significant relationship between plant heights and the number of days to the onset of plant stress ($P = 0.381$). However, significant positive relationship between both spread and height × spread and the number of days to the onset of plant stress was determined (spread: $y = 3.26 + 0.09x$, $R^2 = 15.3\%$, $F = 13.48$, $P < 0.001$; height × spread: $y = 4.76 + 0.001x$, $R^2 = 6.0\%$, $F = 5.40$, $P = 0.023$). Generally, this indicates an unexpected result that species with larger diameter or larger leaf area experienced less stress compared to the species with smaller diameter and smaller leaf area.

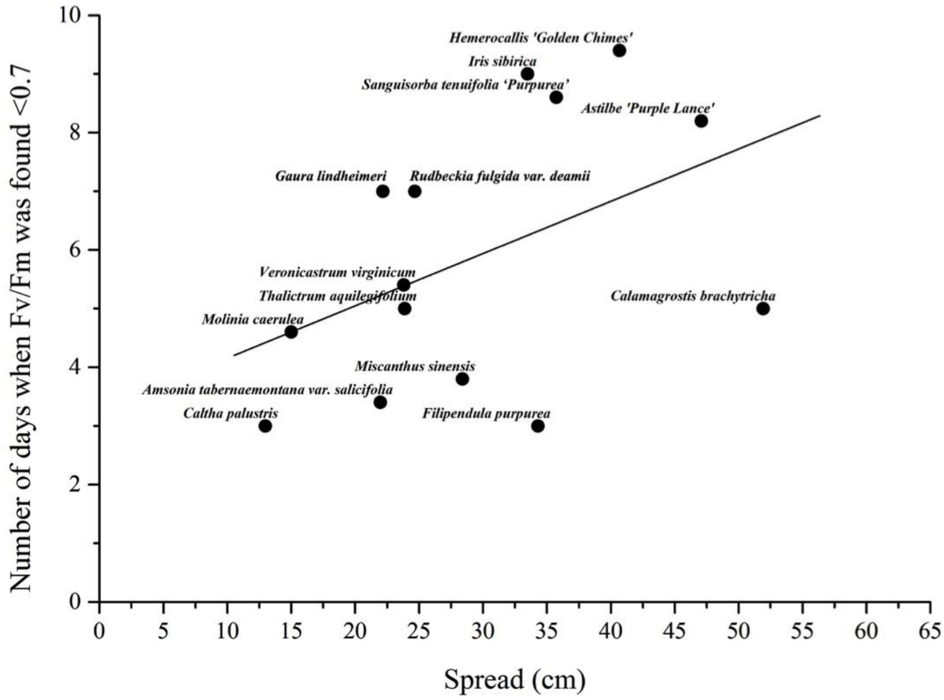


Fig. 3.9. The relationship between plant canopy spread (cm) and the mean value of the number of days to the onset of plant stress (i.e. $F_v/F_m < 0.7$). The fitted line is $y = 3.26 + 0.09x$ ($R^2 = 15.3\%$, $F = 13.48$, $P < 0.001$).

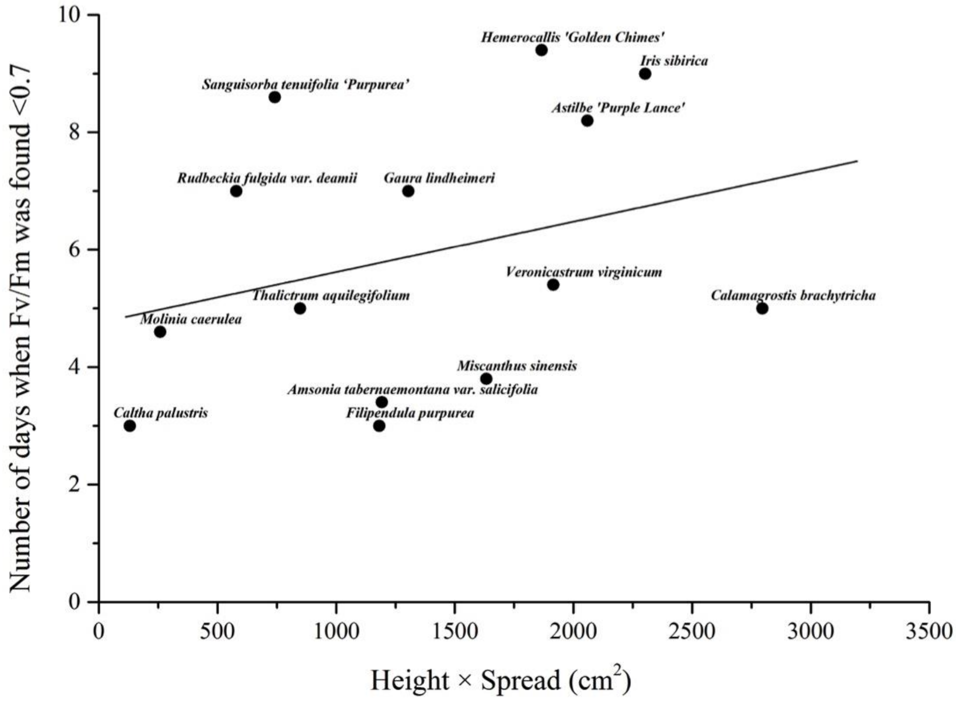


Fig. 3.10. The relationship between plant height × spread (cm) and the mean value of the number of days to the onset of plant stress (i.e. $F_v/F_m < 0.7$). The fitted line is 4.76 + 0.001x (R^2 = 6.0%, F = 5.40, P = 0.023).

Table 3.8 showed the ratings based on the observed performances of survival, physical growths and stress tolerance in individual species. During the whole duration of drought treatment, the selected species showed similar level of drought tolerance according to the result of Friedman test, in which no significant differences between species using all the ratings were indicated (P = 0.362). A league table based on mean rank of individual species' drought tolerances is presented (Table 3.9), though inter-species comparisons indicate that there are no significant differences.

Table 3.8. Ranked drought performances in individual species, including survival, shoot dry weight (SDW), root dry weight (RDW), height (Ht), spread (Spd) and the number of days to the onset of plant stress and soil moisture when $F_v/F_m < 0.7$

Species	Survival	SDW	RDW	Ht	Spd	Number of days to the onset of plant stress	Soil moisture when $F_v/F_m < 0.7$
Amsonia tabernaemontana var. salicifolia	1	4	4	4	4	2	2
Astilbe 'Purple Lance'	3	1	3	4	1	4	4
Calamagrostis brachytrica	3	4	3	2	4	3	4
Caltha palustris	1	/*	/	/	/	2	1
Deschampsia flexuosa	2	3	4	1	1	/	/
Filipendula purpurea	1	/	/	/	/	2	1
Gaura lindheimeri	4	4	2	1	3	4	4
Hemerocallis 'Golden Chimes'	5	3	4	1	3	5	4
Iris sibirica	5	3	4	4	4	5	4
Miscanthus sinensis	3	4	4	1	2	2	2
Molinia caerulea	3	4	4	1	4	2	2
Rudbeckia fulgida var. deamii	3	3	4	1	4	4	4
Sanguisorba tenuifolia 'Purpurea'	4	4	3	2	4	4	4
Thalictrum aquilegifolium	4	4	4	4	4	3	3
Veronicastrum virginicum	3	3	4	4	4	3	3

* *Caltha palustris* and *Filipendula purpurea* had no alive specimen at the termination of experiment, thus had no valid ratings for their physical growths. Chlorophyll fluorescence was not obtained in *Deschampsia flexuosa* due to technical issues, and thus ratings related to stress tolerance were missed.

Table 3.9. League table based on mean rank of individual species' performance in drought treatments. Higher-scored species showed better suitability and overall performance.

Species	Mean Rank
Deschampsia flexuosa	5.50
Miscanthus sinensis	6.00
Caltha palustris	6.64
Filipendula purpurea	6.64
Astilbe 'Purple Lance'	7.07
Molinia caerulea	7.07
Amsonia tabernaemontana var. salicifolia	7.64
Gaura lindheimeri	7.93
Rudbeckia fulgida var. deamii	8.43
Veronicastrum virginicum	8.43
Calamagrostis brachytrica	8.57
Hemerocallis 'Golden Chimes'	9.00
Sanguisorba tenuifolia 'Purpurea'	9.79
Thalictrum aquilegifolium	10.07
Iris sibirica	11.21

3.4 Discussion

3.4.1 Responses to cyclic flooding in selected plant species

100% survival in all the 15 candidate species during the whole cyclic flooding treatments indicates the potential of these species to be considered as appropriate selections for rain gardens. Physiological growths of more than half of the selected species, including *Calamagrostis brachytrica, Caltha palustris, Deschampsia flexuosa, Filipendula purpurea, Hemerocallis 'Golden Chimes', Iris sibirica,* *Thalictrum aquilegifolium* and *Veronicastrum virginicum*, were not affected by cyclic flooding.

The present experiment only applied submergences to substrate level. We therefore assumed that root growth would be the most sensitive of all physiological growth parameters due to the possible damages on root metabolism and nutrient acquisition caused by periodic hypoxia and anoxia resulting from cyclic submergences. The suspicion is proved by the investigation, where the longer-term (4d) interval cyclic flooding significantly decreased the root biomass in *Amsonia tabernaemontana var. salicifolia, Gaura lindheimeri,* and *Sanguisorba tenuifolia 'Purpurea'*. However, it is worth noting that the short-term (1d) interval cyclic flooding tends to have no significant influence on root biomass in all candidate species. Canopy growths in most candidate species possessed adaptive responses to the influence of cyclic flooding treatments, whereas *Gaura lindheimeri* and *Rudbeckia fulgida var. deamii* showed significant decreased canopy spreads in treatments adopted longer-term (4d) interval cyclic flooding. It is also worth noting that cyclic flooding treatments actually promoted the canopy growths of *Astilbe 'Purple Lance'*, *Miscanthus sinensis* and *Molinia caerulea*, which have been suggested to withstand periodic or seasonal inundation in rain garden manuals. Casanova and Brock (2000) concluded similar results that short frequent floods promoted high biomass of two types, including the amphibious species that established their tolerance to the fluctuated inundating-draining process, and those terrestrial species that are capable of growing fast and establishing themselves during the period of draining between floods.

Generally, physiological investigations indicate that the 15 candidate species possessed adaptative responses to the simulated rain garden cyclic flooding conditions. As stated previously, most current technical manuals suggested proper engineering for rain gardens to achieve complete dewatering within 24 hours. All the 15 candidate perennial species in this study showed robust growths in this ideal rain garden moisture regime, as no species had significantly decreased physiological growths due to the short-term (1d) interval cyclic submergence. We therefore recommend suitable engineering for rain garden soil profile to enhance water discharge, so that a wider range of potential species could be considered for use in urban rain gardens.

As previously stated, some species may conserve their growth in vegetative state during flooding to prolong survival or elongate shoots that emerge out of submergence to restore gas exchange. Therefore, species' suitability cannot be simply judged depending on their physiological growths. Chlorophyll fluorescence (i.e. F_v/F_m) as an effective indicator of plant waterlogging stress can provide more insights on predicting the further developments of the candidate species in the expected rain garden moisture profile. Overall, most species maintained relatively good F_v/F_m level during the whole study, which match their fairly good physiological growths, and thus demonstrated their adaptative responses to the experimental cyclic flooding. However, poor health conditions (more than half of the F_v/F_m measurements were found below 0.7) detected in *Amsonia tabernaemontana var. salicifolia* in 4d cyclic flooding group and *Thalictrum aquilegifolium* in both 1d and 4d cyclic flooding treatments, and their limited recovery of photosynthesis efficiency during the draining stages suggest that potential biological injury might occur or become visibly apparent in these plants if more flooding cycles were provided. In contrast, the remarkable recovery of chlorophyll fluorescence in *Calamagrostis brachytrica* and *Caltha palustris* during the flooding stages suggest that the high-frequency stress found in the two species was caused by the given 4-day dry phase for draining between two floodings. We therefore assumed that more flooding cycles might produce less stress in the two species.

Determining the coherent suitability of the candidate species for rain garden moisture profile is necessarily complex. Previous studies often tended to use rather simple methodology and limited indicators to determine species suitability, and may often present counterintuitive conclusions. For instance, Dylewski et al. (2011) found elevated mortality rates, but claimed results were acceptable and plants tolerant of cyclic flooding with no convictive discussion. Furthermore, in this study, there was little reflection as to how the measured physiological growths and health in candidate species may relate to their suitability in typical rain garden moisture profile, while these indicators were simply used to process meaningless between-species comparison based on the unconvincing conclusion (i.e. plants were tolerant choices though elevated mortalities were observed).

In this study, statistical model (Table 3.5) based on their independently scored performances in survival, physical growths (e.g. SDW, RDW, heights and spreads), as well as chlorophyll fluorescence suggest significant between-species differences in their resilience to cyclic flooding treatments. According to the results showed in league table (table 3.5), poor tolerance to rain garden cyclic flooding was determined in *Amsonia tabernaemontana var. salicifolia, Calamagrostis brachytrica, Gaura lindheimeri* and *Thalictrum aquilegifolium*. Poor root and canopy spread growths in *Gaura lindheimeri*, and the weak tolerance to flooding-induced stress in *Amsonia tabernaemontana var. salicifolia* and *Thalictrum aquilegifolium* indicate that the three species are not suitable for longer internal cyclic flooding, and thus should neither be adopted in the frequently damp depression bottoms of rain gardens, nor the slopes with poorly-drained soils in a humid climate. *Calamagrostis brachytrica* had the lowest score among all the candidate species, which is largely due to its poor stress tolerance showed in all the three treatments (i.e. control, 1d and 4d). Plants in this species from 4d group showed the best performance among the three groups, while the recovery of photochemical efficiency in this species actually occurred during flooding stages. The stated facts indicate that this species is water-needy, and may thus not be suitable for the dry rain garden margin and rain gardens in arid regions.

In fact, a considerable proportion of 'banned ornamentals' are wetland species that tend to be invasive in wet environments (Dunnett & Clayden, 2007). In this study, tolerance in *Caltha palustris* was built through the increasing number of flooding cycles, and the poor performance of this species in the control group demonstrate that this species would prefer rather damp condition. Therefore, we consider the potential invasiveness of wetland species would not be problematic to the community dynamics, as adopting wetland species such as *Caltha palustris* in the dryish margins or rain gardens at arid regions with limited precipitation can be very challengeable.

Astilbe 'Purple Lance', *Caltha palustris, Deschampsia flexuosa, Hemerocallis 'Golden Chimes', Rudbeckia fulgida var. deamii, Sanguisorba tenuifolia 'Purpurea'* and *Veronicastrum virginicum* showed fairly good resistance to simulated rain garden cyclic flooding. Most of these species are therefore considered suitable for all the three rain garden saturation zones (i.e. margin, slope and bottom). It is worth noting that F_v/F_m profile showed that photochemical efficiency recovery in *Astilbe 'Purple Lance'* and *Sanguisorba tenuifolia 'Purpurea'* generally occurred in flooding periods, which indicates that the establishments of the two species may demand more moisture, and are therefore considered suitable for rain gardens in a humid climate.

Iris sibirica, Filipendula purpurea, Miscanthus sinensis and *Molinia caerulea* are the highest scored among the 15 candidate species, and are therefore considered suitable for urban rain gardens in a wide range of conditions. It is noticeable that recovery trend of chlorophyll fluorescence in *Iris sibirica* and *Filipendula purpurea* was found during flooding period, which indicates that the two species might prefer to be planted in regions with greater annual rainfall volume.

As stated previously, the mainstream rain garden manuals are North American sources, which often raise the standard recommendations to practitioners to use ‘native-only’ species as they are assumed to be better adapted. However, there is no clear experimental evidence of significant differences of cyclic flooding resistance in the selected species depending on their geographic origins. Adaptation of only native species in urban rain gardens is therefore considered a constraint on diversifying selection of rain garden species.

Results of this study satisfy the hypothesis that the moisture sensitivity of different plant species may fundamentally predetermine their performance and suitability to typical rain garden moisture profile. The suggested spatial distribution throughout the rain garden saturation zones for species with different moisture preferences also generally matches the expectations formed under the conditions mentioned in the current rain garden technical manuals. Overall, species assumed to prefer periodic or seasonal inundation showed the best performances over the species inhabitating the other hydrological regimes. Most species naturally withstand infrequent and continuous inundations were found to have relatively good tolerance in rain garden cyclic flooding conditions. In fact, the performance and suitability of species under the effects of rain garden cyclic flooding condition are also greatly related to their original habitats. There is a general trend that species hail from where moisture excess is available for an extended period of time was higher scored in the league table (Table 3.5). For instance, the highest scored *Iris sibirica* is naturally found in swamps and damp pastures, while *Filipendula purpurea* and *Miscanthus sinensis* naturally grow alongside stream margins or moist lowland meadows where periodic inundation occurs at time. These habitats may often found in regions where high average annual rainfall is guaranteed. In contrast, *Gaura lindheimeri* has been considered as intolerant of inundation by current rain garden manuals as its preference of well-drained sandy loams or chalk, was actually determined as one of the lowest scored candidate species. We therefore assume that moisture preferences in different species would predetermine their distribution, lower limit of biomass and density throughout the typical rain garden moisture gradient ranging from the damp depression bottom to the relatively dry marginal area. In fact, their moisture may often be correctly predicted depending on the habitats where they are found in nature. Nevertheless, it is noticeable that typical rain garden cyclic flooding would greatly challenge the plant health and the planting success of typical moisture-needy perennials naturally growing in wetlands or water boundaries (e.g. *Calamagrostis brachytrica* and *Caltha palustris*) during the draining phrases or the dry periods between precipitations in rain gardens. To sum up, species that withstand infrequent to periodic inundation, especially those of naturally growing in transition zone between upland and wetland (e.g. moist meadows and swamps), are considered the most adaptive species in rain gardens, and are strongly recommended for professionals in future landscape practices.

In this study, the basic ‘pot-in-pot’ methodology was used to simulate the typical interval cyclic flooding conditions occurring in rain gardens. However, it is undoubtedly that the use of container-grown plants would have influence to the experimental observations. Considering the potentially high transpirational water loss due to the elevated temperature in greenhouse during the study and the free-draining soils with limited volume in pots, availability of soil moisture in pot is expected to rapidly decrease during the draining stages and may thus challenge the planting success of some of the moisture-needy species. However, such risks may be weaker in practical rain garden as more soil moisture is expected to be available in planting beds and soils at different depths. Furthermore, plants growing in rain garden bottoms may occasionally encounter deeper flooding to leaf level, and cause direct shading and hypoxia to foliage. However, the effects of deeper inundation to candidate plants were not investigated in the present study as significant loss of substrate was observed when submergence is higher than the substrate level.

Moreover, as stated previously, pots from the control group were maintained at a moisture level of between 0.2 and 0.25 $m^3 \cdot m^{-3}$. The moisture level was suggested in the work of Bailey (2009) and Dylewski et al. (2011) to keep a mesic to moderately moist substrate. However, it appears that they adopted much greater volume of organic component in the medium compared to that we used in this study, for instance, Bailey (2009) used a 9:1 pine bark: sand by volume medium and Dylewski et al. (2011) adopted 1:1 pine bark: peat by volume medium or fine textured calcined clay, whereas we used a sandy textured medium in which half volume was sharp sands. The volumetric water content strongly determined by organic component in the medium, which means the substrates applied in this study may lose moisture easier than that of Bailey (2009) and Dylewski et al. (2011). Therefore, the daily-maintained moisture level of between 0.2 and 0.25 $m^3 \cdot m^{-3}$ may be considered on the dry side and might explain why often control plants did not grow as well as the 1 day flooding.

3.4.2 Responses to drought in selected plant species

In this study, the drought tests for all candidate species were carried out in an extreme scenario, in which irrigation was completely withheld in a greenhouse with elevated temperature during the whole experimental duration. However, as stated previously, typical rain garden, especially those of in the regions with adequate annual rainfall inputs, would rarely encounter extreme dry spell risks. Therefore, drought test in this study is not aiming for simply determining the ordination of tolerance of drought-induced stress in candidate species. It is more important to understand how plant species would respond to drought shock caused by extreme climate fluctuation. In this way, species and corresponding functional group with good drought resistance could be determined and recommended as the "complementary species" for typical rain garden vegetation to increase the community's ability to successfully build good tolerance to prolonged dry spells and maintain their basic ecosystem services.

Mortality was observed in most species from the dry group. Due to irrigation withheld and the noticeable elevated indoor temperature at day times throughout the whole experimental duration, observed mortalities from the dry group were expected and considered acceptable. However, mortality rate may indicate the water use efficiency in specific species to prolong their survivals in face of drought threats, and is closely related to the landscape effect of vegetation. Overall, results of species survival showed that species assumed to withstand infrequent inundation and those of intolerant of inundation had the best survival performance to extreme drought threats. Only *Hemerocallis 'Golden Chimes'* and *Iris sibirica* that have been suggested to withstand infrequent inundation had no mortality at the termination of the experiment. Some of the species that prefer periodic and continuous inundation showed significant mortality rate, such as *Amsonia tabernaemontana var. salicifolia*

(80%), *Deschampsia flexuosa* (60%), and *Caltha palustris* and *Filipendula purpurea* in which no specimens were alive at the termination of the experiment.

It is not possible to make comparison of physiological growths between dry group and control group in *Caltha palustris* and *Filipendula purpurea* due to the complete mortality in the two species. Results obtained from the other candidate species showed that root biomass was the least affected physiological growth parameter that only *Astilbe 'Purple Lance', Calamagrostis brachytrica, Gaura lindheimeri* and *Sanguisorba tenuifolia 'Purpurea'* showed significant decreased root dry weights in dry group compared to the control group. Drought threats plants by causing inadequate soil water availability for plants, thus it is expected that most species would sustain their root expansion to maintain water acquisition. However, it appears that most candidate species could not exploit strategy to respond to the yield decrease of canopy due to drought-induced stress. Plant height was determined as the most sensitive growth parameter to drought, where *Amsonia tabernaemontana var. salicifolia, Astilbe 'Purple Lance', Iris sibirica, Thalictrum aquilegifolium* and *Veronicastrum virginicum* showed significant decreased plant heights. It is worth noticing that the height growth of plant shoots may closely related to flower production of herbaceous species, where reduced shoot height growth would severely increase the days to flower initiation and reduce the length of flowering display (Ollerton & Lack, 1998; Sun & Frelich, 2011), and consequently restrict their aesthetics. We therefore suggest that using species had significant reduction in shoot heights should be cautious for their potentially worse aesthetic values compared to the other tolerant species.

Only two species including *Amsonia tabernaemontana var. salicifolia* and *Thalictrum aquilegifolium* had no significant drought-induced reduction in physical growths, while the intolerance levels demonstrated by decreased physiological growth in the rest of candidate species were varied. Physiological growth performance in a few species could not match their survival outcomes. For instance, the fair drought tolerance in *Amsonia tabernaemontana var. salicifolia* could not be simply assumed by its non-affected physiological growth due to the noticeable mortality (80%) found in this species. In contrast, dry shoot weight and canopy height and spread of *Hemerocallis 'Golden Chimes'* were significantly reduced due to drought treatment, however this species show no mortality at end of the experiment. Therefore, between-species suitability to extended drought could not be simply determined by mortality rate and physiological growth obtained at the termination of the experiment.

Chlorophyll fluorescence investigation also provides important insights for evaluating the drought resistance in candidate species. The number of days on the onset of stress (i.e. $F_v/F_m < 0.7$) is an indicator of the sustainability of species encountering extreme dry spell. Species showed a greater number of days before being affected by drought showed their abilities to prolong survival during an extended drought period, and thus may help maintain the community in practical rain garden till the next rainfall event. We also consider the lower soil moisture in which the onset of drought-induced stress was observed presents greater water use efficiency in species. Investigation shows that most candidate species showed high water use efficiency also tended to maintain their health for relatively long time in the extreme dry spell.

In general, chlorophyll fluorescence investigation not only shows that species with different moisture preferences have different drought-induced stress levels, but also demonstrates a trend which matching the conclusion derived from the survival investigation. Plant species assumed to withstand infrequent inundation showed the best drought resistance amongst the other groups. Measurements of chlorophyll fluorescence showed that *Hemerocallis 'Golden Chimes'* and *Iris sibirica* had the longest tolerance duration over the other species before they exhibited drought-induced stress, whilst *Sanguisorba tenuifolia 'Purpurea'* was found under stress at the lowest level of soil moisture. *Gaura lindheimeri* assumed to be intolerant of inundation also exhibited stress at the lowest soil moisture level amongst others. However, the resistance duration before the onset of drought-induced stress in this species was shorter than in *Hemerocallis 'Golden Chimes'*, *Iris sibirica*, *Sanguisorba tenuifolia 'Purpurea'*, and *Astilbe 'Purple Lance'*. Experimental observations in *Caltha palustris* assumed to withstand continuous inundation suggest that this species had extreme low water use efficiency, and could only maintain its health for the shortest duration amongst the candidate species. The F_v/F_m profile suggests that plant species assumed to withstand periodic or seasonal inundation had variable drought resistance levels. Most species from this group showed an ordinary level of drought tolerance. However, rather poor drought resistance similar to that of *Caltha palustris* was observed in *Filipendula purpurea*. In contrast, *Astilbe 'Purple Lance'* showed an extraordinary drought tolerance, which exhibited stress after a fairly long duration in drought conditions and endured rather low soil moisture availability.

In extended drought conditions, how quick the plants would die depending on how quick the moisture leave their foliage. Water losses in plants through evapotranspiration are closely related to the vapour pressure deficit (VPD) (Allen *et al.*, 1998). However, a limitation in this experiment was that the VPD was not formally measured to understand the drying power of the air and its impact on plants, which is thus suggested for the future work. Leaf area is also expected to greatly influence the stress tolerance across species. It is assumed that species with larger size or leaf area would have a promoted evapotranspiration and thus dries out quickly and experienced more stress compared to the species with smaller size or leaf area. However, it is surprisingly that in this study, results show that species with bigger leaf area generally experienced less stress than species with smaller leaf area. The major concern is that some of the bigger species used in this study may be naturally more drought-tolerant compared with the smaller species. For example, *Hemerocallis 'Golden Chimes'* with much bigger leaf area maintained its health for a much longer period of time compared to *Caltha palustris*. The former is normally found in moist but well-drained soil, whilst the latter is a typical wetland species that hails from wet areas in marshes and ditches.

Similar to the situations in the cyclic flooding treatments, the coherent analysis depending on candidate species' mortality rate, physiological growth and stress levels showed no clear evidence to support the native/non-native argument of differences in drought tolerances in species from different geographical origins. It also suggests that there is no significant between-species difference in drought tolerance. However, we assumed that the significance would be determined if there were more samples in the tested population. In fact, the league table depending on the mean rank of individual species' performance (Table 3.9) shows a relatively obvious trend that the water-needy species adopted from wet and damp habitats tended to have the worst tolerance to extended drought, whereas those requiring only infrequent inundation tended to have better adaptive response to drought-induced stress. This trend also match the conclusions derived from the results of survival and chlorophyll fluorescence investigation. It is worth noting that although the assumed drought-tolerant *Gaura lindheimeri* showed very good survival and stress tolerance, this species only had average score amongst the candidate species due to its significantly decreased physiological canopy growth, especially of the stunted height growth.

As stated previously, adding drought tolerant species would not only strengthen the community's resilience to weather shocks, but also help maintain the visual appeal of rain garden vegetation in the extreme dry spell. Species prefer infrequent inundation showed the best overall performance in both cyclic flooding and drought treatments, and are therefore strongly recommended for future practices. The experimental observation does not mean that species showed poor drought tolerance is not suitable for rain garden moisture profile. We suggest that species intolerant of extended drought should be planted in moist to damp rain garden bottom and slope, and avoid adopting in the dry marginal areas and the well-drained slopes in arid regions.

It is worth noting that containerised plants often tend to develop larger canopies than naturally regenerated seedlings (Tsakaldimi et al., 2009), therefore, transpirational water losses from container-grown plants are expected to be greater than vegetation in practical rain gardens. Moreover, lateral spreads and length of roots could be severely restricted within containers, and no roots of all the experimental specimens were able to grow out from the base of the pots during the whole study, whereas plants in practical rain gardens are expected to have more extended root systems. As stated previously, species with extended roots could reach different depths is soil where adequate moisture is available to prolong survival in an extreme dry spell. Therefore, we assume that most of the candidate species would show better sustainability in practical rain gardens during dry spell, especially those deep-rooting species such as *Miscanthus sinensis*.

3.5 Conclusion

Plant health plays a major role in maintaining the functionality and aesthetics of rain gardens, therefore rain garden successes dependent on proper species choice, where. However, the current rain garden application and researches on plant suitability in typical rain garden soil moisture profile often tend to be two isolated processes. It appears that landscape architects may adopt species without being previously tested in scientific experiments or suggested by technical manuals. On the other hand, previous studies often showed little species diversity from rather limited range of selection, and may adopt incorrect methodology leading to inappropriate suggestions.

This study represents an important step in understanding the influences of typical cyclic flooding and potential extreme drought scenarios in rain gardens on plant establishment. Adapting the measurements of physical plant growth parameters such as shoot dry weight (SDW), root dry weight (RDW), height and spread coupled with the stress indicator (i.e. chlorophyll fluorescence) can help identify tolerant species and ecotypes for the dynamic rain garden soil moisture conditions. This study is thus valuable for guiding future collaborative research and application to choose appropriate species that are likely to be suited to life in given saturation zones that are subject to differing soil moisture conditions throughout the depression structure of the rain gardens.

Overall, this study suggests that species have been suggested to withstand infrequent to periodic inundation not only showed the best adaptation to rain garden cyclic flooding amongst all candidate species, but also are considered the most adaptative to most saturation zones throughout the moisture gradient in rain gardens. Species preferring infrequent inundation also showed the best tolerance to extreme dry spell, and is therefore considered the most appropriate choices for the expected rain garden moisture profile in a wide range of climate conditions. Furthermore, moisture sensitivities of plant species are closely related their original habitat, so that we suggest landscape practitioners to propagate species from transition zones between damp lowland and dry upland and those of moist but well-drained soils.

No experimental observations could support the native/non-native debate on plant hardiness and suitability in rain garden soil moisture profile, so that the ‘native only’ rule is considered a constraint for plant selection. As stated previously, there are many potential reference communities from corresponding habitats that may be successfully adopted in rain gardens to increase the functional diversity in urban living environment. The evaluation and adaptation of potential wild reference communities and providing guideline for professions is therefore considered an important direction for future research.

The 24-hour interval cyclic flooding showed no significant effects on most candidate species in this study, and was determined to promote the physiological growths of a few moisture-needy species. Completely dewatering within 24 hours is an ideal standard for soil engineering in urban rain garden and is therefore strongly recommended. As a result, the potential species that are intended to be planted in rain gardens will remain healthier and live longer, thereby reducing the costs for maintenance and labour.

Leaf chlorophyll fluorescence was used as an indicator for evaluating waterlogging/drought tolerance in the candidate plants. It was found easy to use and very effective to reflect plant stress caused by typical flooding cycle and potential drought in rain gardens. This method required less time to reveal the waterlogging/drought tolerance in plants compared to the traditional method of measuring the physical growth of plants and was less destructive to plants, but was also able to reveal the invisible biological damages in plants to predict stress. It is thus highly recommended for use in future research.

In this study, soil moisture condition was the most important controlled factor that greatly influenced the growth and health conditions of selected plants, however, temperature and air humidity were acknowledged to be potential limitations with the experiments that altered the plant stresses in a few species. It is thus recommended that future research shall be carried out under a stable range of temperature and air humidity. Tested perennials in control, 1d and 4d groups reached their maximum growths in greenhouse with an elevated temperature, this experiment terminated in a short duration (i.e. one month). However, species could become increasingly tolerant of flooding as plants mature (Middleton, 2002), thus results of most species could predict their further adaptions to the cyclic flooding treatments.

In real rain garden conditions, plants may experience weather shocks such as moving rapidly from drought to flood or the reverse. It is valuable to design a controlled condition with a repetitive cycle that rapidly switches between drought (extremely low soil moisture) and flood (inundation) to know how the plants cope with weather shocks and to identify suitable species for extreme conditions. Additionally, it is also very valuable to know how periodical deeper inundations (e.g. flood to leaf level in plants) affect the plants in future research. This was not done because of time constraints and because the loss of substrates from pots in deeper inundation over substrate level could not be solved, and is thus expected to be investigated in practical rain gardens in the future.

Chapter 4: Hydrologic performance of a taxonomically diverse forb-rich plant community in rain gardens

4.1 Introduction

4.1.1 Issues of traditional planting in rain gardens and the introduction of a taxonomically diverse forb-rich plant community

Previous studies on climate suggest that the frequency of high intensity storms and rainfall intensity will increase in the UK (Jones et al., 2013; Kendon et al., 2014). Therefore, reducing the amount and rate of runoff is an important consideration in managing the effects of heavy rainfall in built-up areas, where highly disruptive flash-flooding is becoming an increasing problem. Recent studies indicate that stormwater management facilities such as rain gardens could greatly reduce the amount of stormwater runoff compared to hard surfaces in a variety of climates (Burge et al., 2012). Rain gardens are sited close to, or adjacent to buildings, roads, pavements and car parks, to retain rainfall on site, and return it to the ground water or to the atmosphere, and reduce the amount of runoff leaving impervious or sealed urban surfaces. These features rely heavily on the role of vegetation and soils to capture and clean excess rainwater runoff in urban areas and return them to the atmosphere (Dietz, 2007; Dunnett & Clayden, 2007; Schroll et al., 2011).

In many cases, rain gardens are in the public right-of-way, and are therefore the perfect situations for introducing very attractive plantings, which add biodiversity and aesthetic value into areas that would otherwise be devoid of vegetation. However, the importance of planting design and vegetation technologies has often been underestimated in the implementation of rain gardens. Many of these measures are found dominated by vegetation with low species richness compositions (Fig. 4.1), which lead to unnatural and sometimes unpleasing visual performances of plantings that are very simple, or result in a poor interaction with local biodiversity (Dunnett, 2004). Close-cropped mown vegetation (e.g. mown grasses) is also likely to occur in rain gardens, which could turn into a muddy quagmire and may sometimes be removed altogether leaving nothing but bare soil (Steiner & Domm, 2012). The removal/loss of vegetation considerably reduces rain garden's ability to hold soil in place, thus leading to erosion and reduced contribution to stormwater infiltration and filtration (Steiner &Domm, 2012).

Fig. 4.1. Street rain garden planted with shrub monoculture featuring a rather boring visual display and low biodiversity. Columbus, USA. Photo was taken by the author in October 2012.

To cope with these issues in planting of rain gardens, the use of taxonomically diverse mixes of forb-rich plant communities are proposed as an alternative vegetation approach. The taxonomically diverse forb-rich plant communities emphasise the structurally complex plant communities with high plant diversity and phenological changes to ensure visual interest over time, and which only require minimal input for implementation and a minimised environmental impact (Hitchmough & Dunnett, 2003). Vegetation such as the forb-rich meadows and prairies have been progressively applied in the urban context and received positive agreement among the public (Kingsbury, 1996). The taxonomically diverse mix of a forb-rich plant community is also becoming a desirable vegetation technology for rain gardens (Dunnett & Clayden, 2007). The use of such vegetation is emerging in the UK and the US. For example, some domestic rain gardens in the US and the UK have adopted the border like plantings, which are strongly valued for their aesthetics with various colours in flowering displays (Fig. 4.2) (Steiner & Domm, 2012). There are also some good examples found in other stormwater management facilities such as green roofs. For instance, the green roof installed on the Moorgate Crofts building, Rotherham, UK, had successfully demonstrated the aesthetic and social values and rainwater reuse function provided by a semi-extensive mixture of alpines and sedums requiring minimal maintenance (Dunnett & Clayden, 2007) (Fig. 4.3). Using diverse and attractive forb-rich vegetation can also provide further benefits beyond stormwater management, such as biodiversity conservation (Kazemi et al., 2009; Kazemi et al., 2011) and aesthetic amenity (Dunnett & Clayden, 2007, Hitchmough & Wagner, 2013).

Fig. 4.2. A domestic rain garden featured a taxonomically diverse border planting with flowering meadow species, Sheffield, UK. Photo was taken by the author in July 2013.

Fig. 4.3. Semi-extensive mix of alpines and sedums applied in a green roof, Moorgate Crofts, Rotherham, UK. Photo was taken by the author in August 2013.

4.1.2 Mechanism of the quantitative control of stormwater runoff in rain gardens

Rain gardens have the same ecological processes to cope with stormwater as other stormwater management facilities such as green roofs and retention basins, which are highly dependent on the role of vegetation and soils to store, infiltrate and evaporate stormwater, thus reducing the runoff in urban areas. Retention (i.e. the reduction of the amount of rainfall that becomes runoff) and detention (i.e. the lag and attenuation of the runoff hydrograph) are the two major stormwater management effects that contribute to the control of stormwater runoff quantity in an urban area (Stovin et al., 2015). Retention and detention parameters (e.g. runoff peak attenuation and quantitative descriptions of lag time such as the time to onset of runoff, runoff peak delay, centroid delay and t_{50} delay) are the most popular indicators of the systems' hydrological performance (Stovin et al., 2012; Stovin et al., 2015). Fig. 4.4 shows sketches of rainfall and runoff hydrograph in a green roof setting adapted from Stovin et al. (2015) for illustrating the retention and detention metrics. Peak attenuation refers to the reduction in peak flow rate of runoff compared to that of the rainfall (Fig. 4.4b) (Stovin et al., 2012). Centroid delay is determined from the time between the centroids (the centre of mass) of the water inflow and that of the runoff outflow (Fig. 4.4b) (Leopold, 1991; Stovin et al., 2012). t_{50} delay refers to the time difference between the 50% value on the cumulative rainfall profile and the same absolute depth on the cumulative runoff profile (Fig. 4.4c) (Stovin et al., 2015).

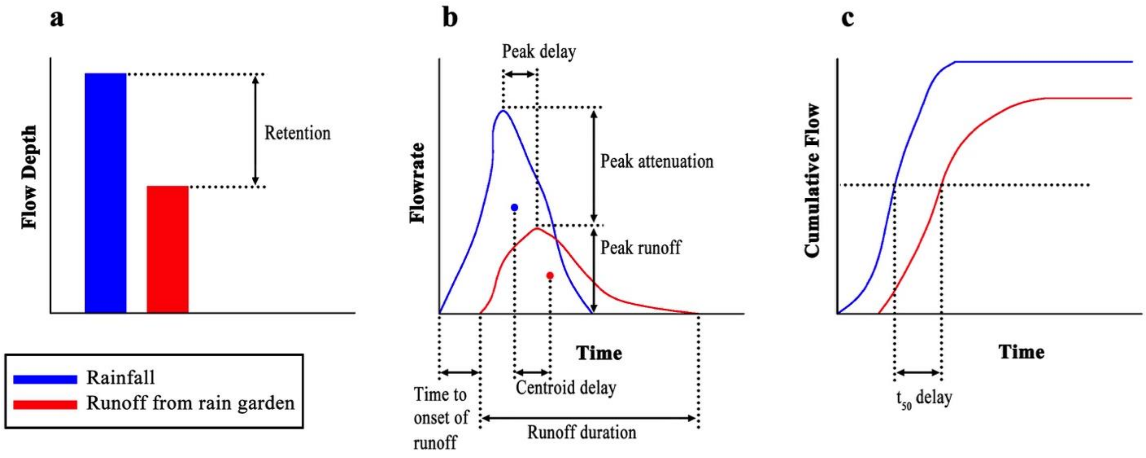


Fig. 4.4. Retention and detention metrics (Schematic) (4b and 4c are adapted from Stovin et al., 2015).

During a storm event, rain gardens retain a certain portion of stormwater until it reaches the soil field capacity. These initial losses may be compared to the immediate runoff from the impervious surfaces. The retained water will eventually return to the ground via infiltration (i.e. the downward movement of rainwater through soil) or to the atmosphere via evapotranspiration (Jennings et al., 2015). A certain volume of the retained water could be absorbed into micropores in soils and taken up by the absorbent materials in the substrates (Bengtsson et al., 2005). Increasing the soil porosity helps the adsorption of rainfall on-site, thus the soil infiltration is improved (Dietz & Clausen. 2006).

Evapotranspiration is governed by the direct loss from evaporation from bare soils and pooled water, and would be greatly increased in vegetated areas through the contribution of transpiration from plants (Allen et al., 1998; Dunnett & Clayden, 2007). Heat is the vital energy needed for evaporation, so that the evapotranspiration in rain gardens and other sustainable stormwater management components can be greatly affected by climate, where the evapotranspiration would be reduced in cooler climates (Hunt et al., 2006; Poë et al., 2015). However, in practice, the contributions between the two simultaneous processes of evaporation and transpiration to the combined term of evapotranspiration could hardly be distinguished from each other (Ward & Trimble, 2004). It is also worth noting that the effectiveness of retention in practical rain garden is dominated by infiltration, while evapotranspiration is reported to play minor roles in runoff reduction compared to the loss through infiltration (Jennings et al., 2015). However, many related studies often tend to not properly measure evapotranspiration and simply assumed the major retention in rain gardens and stormwater management facilities was the loss through evapotranspiration.

Furthermore, Stovin et al. (2012) and many others suggested that a longer antecedent dry weather period (ADWP) might result in higher retention per storm event than a shorter ADWP. The meteorological conditions associated with the ADWPs vary the evapotranspiration rates seasonally or daily by changing the recovery rate of stormwater management facilities' retention capacity. For instance, a faster retention capacity recovery could be expected in the summer with a higher evapotranspiration rate than in winter (Hunt et al., 2006).

Detention occurs because it takes time for rainwater to flow through the soil media depth and drainage layer before it converts to runoff (Stovin et al., 2012), thus a time lag between the runoff peak from a pavement area and a rain garden in the same rainfall event is generated. For instance, Lloyd et al. (2002) reported a longer period of time lag, up to 30 minutes, for the discharge of stormwater runoff in a rain garden setting than in the conventional piped systems. Detention also decreases peak runoff outflow from rain garden systems. For example, the study of Lu and Yuan (2011) suggested significant peak attenuation in rain garden that decreased approximately 95.6% peak flow (from 84110.4 cm^3/s to 3681.6 cm^3/s).

To date, most studies focused on the evaluation of runoff drainage in soil amendments and the whole systems of stormwater management facilities from an engineering point of view (Cheah & Ball, 2007; Kronaveter et al., 2001; Yang et al., 2009). Previous research mainly focused on how well the stormwater management facilities could achieve better retention for urban contexts (Nagase & Dunnett, 2012; Stovin et al., 2012; Yang et al., 2009). However, there is a lack of consistency in published hydrological performance data of the stormwater management facilities, particularly concerning detention performance. Stovin et al (2012) suggested that runoff happens almost instantaneously with rainfall unless there is retention capacity still remaining, and the detention effect should be independent of retention and should only relate to the delay experienced by the runoff once the substrate has reached field capacity. Many previous studies presented inappropriate data of runoff delay due to the retention of the first part of the rainfall event rather than a result of detention, and thus suggested apparent lag effect on runoff and very long detention duration in stormwater management components (Stovin et al., 2015). Testing detention independently of retention requires the use of a pre-wetted system (Stovin et al., 2015).

4.1.3 Influence of plants for the stormwater quantitative control in rain gardens

Plants are suggested to have the ability to alter the soil-water dynamics in various ways. Stormwater in stormwater management facilities could be temporarily stored in plant tissues, which means that the volume of runoff is reduced (Nagase & Dunnett, 2012). Virahsawmy et al. (2014) reported that the infiltration in rain gardens adopting vegetated media could be approximately 150 mm/h higher than fine sandy medium without vegetative treatments, which highlighted the great role of vegetation in improving soil permeability. Growth of roots can reverse soil compaction (Yunusa & Newton, 2003), as well as to create, enlarge and elongate soil pores following their root turnover (McCallum *et al,* 2004; Ball *et al,* 2005). In this way, soil hydraulic conductivity and infiltration capacity are maintained over time to govern retention capacity.

Rain garden retention would generally vary by seasons, which may be promoted through accelerated loss of water back to the atmosphere caused by increasing available solar energy and transpiration from plants in canopy growing seasons (Lundholm et al., 2010). It is fairly well established that the greater vegetation aboveground growth traits (i.e. canopy height and diameter, dry weight of shoots) could increase the evapotranspiration (Vertessy et al., 1995) and the reduction of the runoff quantity in stormwater management facilities (Nagase & Dunnett, 2012). For instance, Hunt et al. (2006) installed a practical rain garden planted with *Betula nigra*, *Juncus effuses*, *Iris pseudacorus* and *Magnolia virginiana* at one plant per 4 m^2 was conducted in Greensboro, North Carolina, USA, which was sized to be 5% of an approximately 2000 m^2 impervious drainage area. From June 2002 to May 2003, on-site precipitation and runoff outflow from the underdrain pipe installed in the rain garden were instrumented, respectively. During the experimental period, 78% of the runoff from over 48 observed rainfall events was reduced by the installation of the rain garden, while significant seasonal variation was observed as 86%, 93%, 87% and 46% during spring, summer, autumn and winter, respectively. The noticeable seasonal variation in runoff reduction from rain gardens is hypothesised to be caused by the varied temperatures in different seasons, and the transpiration from plants and the canopy growths are greater in the growing season compared to the non-growing season (e.g. in winter when deciduous plants have no leaves or have been closely clipped).

Different vegetation types adopting different structures and growth habits may alter the hydrological budget and hydraulic performance of rain garden’s infiltration system. Culbertson (2008) and Barrett et al. (2013) concluded that plants have taller and larger aboveground traits and deep roots promoted the evapotranspiration to help the sustainable stormwater systems to retain more runoff. Barrett et al. (2013) and Gonzalez-Merchan et al. (2015) suggested that thicker roots and greater root extension through sediments are more appropriate to limit soil clogging, whereas some species are likely to accumulate sediments in their root balls forming impervious buffer could lead to less effective hydraulic performance in rain gardens.

Moreover, Rixen & Mulder (2005) suggested that there is a parallel increase in runoff retention and species richness due to the greater diversity of structures (mainly aboveground configurations) in high-diversity vegetation. Dunnett et al. (2008) also demonstrated that vegetation composition might affect the dynamics of runoff amounts in stormwater management facilities, so that vegetation with high species richness might achieve better hydrologic response to stormwater inputs. However, experimental studies and solid data on the effectiveness of the recommended taxonomically diverse forb-rich vegetation mixes have not yet been explicitly provided. A few limited studies have shown varied results on the rain garden’s stormwater management performance with different vegetation types including the taxonomically diverse plantings such as meadows and prairies and the conventional vegetation with low species richness and structural diversity (e.g.

mown grasses). Johnston (2011) demonstrated that: (a) the prairie treatments in model rain gardens displayed significantly higher runoff retention than the residential turfgrass treatments and bare-soil which were not different from one another in small rainfall inputs (10-50 mm), (b) the prairie had the greatest runoff retention, followed by turfgrass and bare-soil, in the medium size precipitation (51-90 mm), (c) the prairie and turfgrass treatments displayed runoff retention which were not different from one another, but were significantly greater compared to non-vegetated soil in the large rainfall inputs (91-130 mm). Johnston (2011) suggested that evapotranspiration during periods of dry weather and the plant-induced soil structural development may explain the differences in soil-water dynamics from different vegetation types: (a) prairie and bare-soil had less antecedent soil moisture (i.e. higher soil water transpiration) over the turfgrass treatments at soil surface (0-0.15 m depth), where the antecedent soil moisture ranked from least to greatest in order of the prairie, turf and bare-soil at rooting zone (0.30-0.45 m depth), (b) soil in turfgrass and prairie treatments exhibited infiltration rates that were not different from one another, but were significantly greater than from the bare-soil, (c) vegetation showed a reversal to the soil compaction with increased microporosity resulting in larger soil hydraulic conductivity from prairie and turfgrass treatments than from the bare-soil, (d) turfgrasses had greater soil hydraulic conductivity than in prairie at the soil surface (0-0.05 m), however, significantly less hydraulic conductivity was found in turfgrasses than in prairie at 0.30-0.45 m depth. However, Lundholm et al. (2010) found a conflicting result that bare soil captured more runoff compared to vegetative treatments. Nevertheless, Nagase and Dunnett (2012) and Dunnett et al. (2008) demonstrated that grasses would retain a greater amount of runoff than forbs in stormwater management facilities. The contradictory results might be caused by different experimental design and experimental environments, substrate types, differences in the amounts of rainfall added and the nature of plant traits. Furthermore, a concern for all the three stated studies is that the effect of ADWPs on the runoff retention in these tested sustainable stormwater management components was not discussed.

4.1.4 Research objectives

Concerns of the remaining unclear effects of vegetation type, especially for the designed taxonomically diverse mix of forb-rich plants and the traditional vegetation with low species richness and structural diversity (e.g. mown grasses) on the retention and detention in rain gardens leave important research gaps, and are therefore the focus of this study. This study therefore seeks to experimentally investigate the effects of the proposed taxonomically diverse mix of forb-rich plantings on the hydrology of model rain gardens to improve this functional understanding and determine the differences of retention and detention between the taxonomically diverse mix of forb-rich plantings and the conventional mown grasses. The objectives of this experiment comprised:

(1) To quantify the effect of plant diversity (taxonomic and structural diversity) on the retention response of model rain gardens receiving applications of artificial rainfall inputs under different ADWPs.

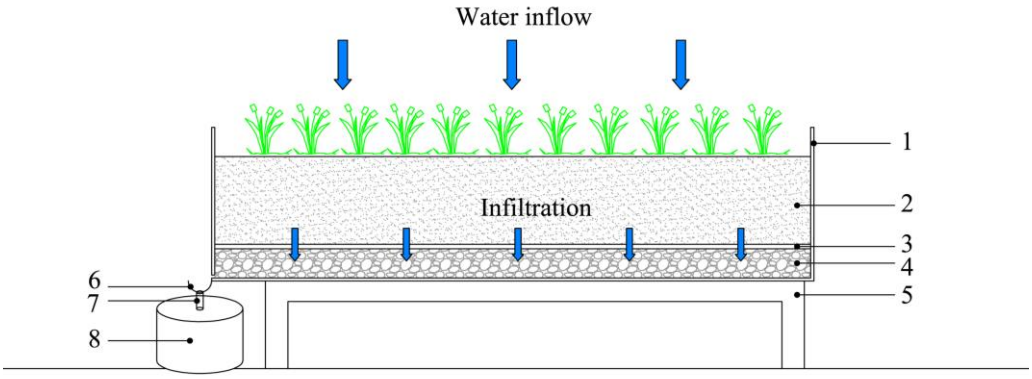
(2) To correctly identify the detention effects of different vegetation types using pre-wetted systems.

4.2 Methods

In this study, hydrological measurements were collected from experimental rain garden modules planted with typical short amenity mown grasses (low species richness and structural diversity) and a mix of taxonomically diverse forb-rich plant community (high species richness and structural diversity) and a control group with bare soil, to address the effect of the vegetation type and the differences in the antecedent dry weather period (ADWP) on the hydrologic dynamics of rain garden. The experimental rain garden modules experienced (a) simulated rainfall inputs equal to a 1 hour storm event having a 10-year recurrence interval for testing retention characteristics under three different ADWPs (2 days, 5 days and 7 days), and (b) an hour-long application of artificial rainfall with pre-wetted soil media (i.e. irrigate it to its field capacity so that all inflow becomes runoff) for testing detention characteristics. Details of the selected rainfall applications are provided in section 2.2.

4.2.1 Site and materials

This study was conducted at a nursery area located in Green Estate Ltd., Sheffield, UK (1˚26’11‘‘W, 53˚22’37’‘N). Fifteen experimental rain garden modules were constructed on site using uncovered plywood boxes with a surface area of 2000 mm by 1000 mm (2 m^2) and a depth of 500 mm to form their structure. The inside space of these boxes were covered by impervious liners to form the ponding structure and ensure that no water can escape from the joints of the modules (Fig. 4.5). All rain garden modules were filled with the same soil mixture as that used in the previous study of this PhD (Chapter 3). The soil media was a mixture of sharp sand, topsoil and compost (5:2:3, volume ratio) and was classified as a gritty sandy loam (67.2% sand, 13.7% silt and 0.01% clay) with an organic matter content of 8.21% and a pH of 7.9. The substrate was free-draining with a porosity of 66.5% and a drainage rate of 5.7 cm/hour, which is ideal for the water infiltration and planting establishment in typical rain gardens (it was recommended from 0.25 cm/hour in clay loam soil to 21 cm/hour in sandy loam soil, Woelfle-Erskine & Uncapher, 2012). Each module had a 100 mm ponding depth. A drainage layer (depth 100 mm) was placed at the base of each module. It comprised ~20 mm pea gravel and was separated from the overlaying soil medium (depth 300 mm) with a filter mat to prevent the drain from clogging (Fig. 4.5). The bases of the modules were laid at a slope of 1.5˚. Each module had a water outlet port (1000 mm ×10 mm), so that the runoff outflow could leak into a gutter then be diverted to the water tank for measurement (Fig. 4.5). The experimental modules were oriented adjacent to one another in a line on open flat ground, with 400 mm spacing between each module. The modules were supported 300 mm above the ground to provide room beneath for the runoff collection tanks and to allow easy access for monitoring (Fig. 4.5).



1. Impervious linear 2. Growing media (300 mm depth)
3. Filter mat 4. Aggregate drainage layer (100 mm depth)
5. Supporting frame (300 mm height)
6. Gutter 7. Discharge pipe 8. Runoff collection tank

Fig. 4.5. Sectional illustration of the experimental rain garden modules

To test the effects of vegetation type on the retention and detention of these systems, different vegetation treatments were used. The experimental systems had three treatments: mown grasses (A series), mixed forb-rich perennials (B series) and a non-vegetated control group. Each treatment had five modules as replications. The mown grasses in A series consisted of a commercial mixture of six typical lowland grass species, which was obtained from Emorsgate Seeds Ltd. (Norfolk, UK). The selected grass species included *Agrostis capillaris, Alopecurus pratensis, Anthoxanthum odoratum, Cynosurus cristatus, Deschampsia cespitosa, Festuca* *rubra*, and their characteristics are shown in Table 1. The mixed forb-rich perennial plantings in B series consisted of eight forbs and two grasses (Table 4.1). The selected species in the B series were ten desirable species from the previous study in this PhD (Chapter 3), which included *Amsonia tabernaemontana var. salicifolia, Astilbe 'Purple Lance', Calamagrostis brachytricha, Filipendula purpurea, Hemerocallis 'Golden Chimes', Iris sibirica, Molinia caerulea, Rudbeckia fulgida var. deamii, Sanguisorba tenuifolia 'Purpurea', Veronicastrum virginicum.* These selected species were identified as not only showing good adaptation to the cyclic flooding conditions in typical rain gardens, but as also having fairly good tolerance to potential drought.

Table 4.1. Characteristics of the 16 plant species in mixtures and their ecological and morphological characteristics (Brickwell, 2008; Hubbard, 1984)

System category	Species	Proportion[a] in mixture (%)	Plant type	Leaves
A	***Agrostis capillaris***	15	Grass	Tufted; erect or spreading culms; slender and hairless leaves
	Alopecurus pratensis	6.25	Grass	A loosely or compactly tufted; culms are erect or kneed at the base; hairless leaves are slender to moderately stout
	Anthoxanthum odoratum	1.25	Grass	Tufted; unbranched culms are erect or spreading, slender to relatively stout; leaves loosely to densely bearded at the apex, otherwise smooth or loosely hairy
	Cynosurus cristatus	45	Grass	Compactly tufted; stiff unbranched culms erect or slightly spreading; hairless smooth leaves
	Deschampsia cespitosa	1.25	Grass	Densely tufted, high and forming large tussocks; culms erect, moderately slender to stout; hairless leaves
	Festuca rubra	31.25	Grass	Tufted, high, relatively long to very long slender; culms erect or curved towards the base, slender to relatively stout; hairless leaves
B	***Amsonia tabernaemontana* var. salicifolia**	10	Forb	Clump-forming with numerous stems; small, ovate to elliptic or lance-shaped leaves
	***Astilbe* 'Purple Lance'**	10	Forb	Tall, vigorous, densely clump-forming and rhizomatous; ovate handsome leaves with each leaflet further divided into 3-5 toothed lobes
	Calamagrostis brachytricha	10	Grass	Tufted, clump-forming rather large ornamental grass; culms erect; long leaves
	Filipendula purpurea	10	Forb	Clump-forming; bearing pinnate, toothed leaves, with irregularly 5 to 7 lobes, rounded to obovate terminal leaflets
	***Hemerocallis* 'Golden Chimes'**	10	Forb	Evergreen; slender, well-branched, reddish brown scapes; narrow leaves
	Iris sibirica	10	Forb	Rhizomatous, beardless; narrow, grass-like leaves

Molinia caerulea	10	Grass	Compactly tufted, high, often forming large tussocks; culms erect, slender to somewhat stout; smooth hairy leaves
Rudbeckia fulgida var. deamii	10	Forb	Rhizomatous; very hairy stems; long-pointed, ovate or oval-ovate, toothed, rough basal and stem leaves
Sanguisorba tenuifolia 'Purpurea'	10	Forb	Slender and erect stems; narrow serrated foliage
Veronicastrum virginicum	10	Forb	Erect, hairless and unbranched stems; lance-shaped to inversely lance-shaped, pointed, toothed leaves, in whorls of 3-7

a: In mown grasses (A series), proportion is described as the proportion of the weight of the total seeds of each species in the seed mixes. In mixed forb-rich perennial plantings (B series), the proportion means the proportion of the number of total plants of each species in the mix.

b: Mown grasses (A series) were mown monthly to maintain the height of 10 cm.

Seeds of grass mixes in A series were sown into the modules on 22 April 2013 at the sowing rate of 2 g/m^2 (which was the minimal sowing rate suggested by Emorsgate Seeds Ltd., Norfolk, UK, compared to the normal sowing rate at 20-25 g/m^2 to create amenity grass), and were then allowed five months to establish before the experiments were conducted. From July 2013, monthly hand-shears were undertaken in the grass mixes to maintain their heights at approximately 10 cm. Modules in B series were planted with young plants supplied in 9 cm diameter pots from 28 to 30 April 2013, which were ordered from Orchard Dene Nurseries (Lower Assendon, Henley-on-Thames, Oxfordshire, UK). The plants in B series were initially planted on a grid of 200 mm spacing intervals based on their growing habits. The non-vegetated control modules were raked monthly to maintain flat ground. All modules were maintained by hand-weeding and no supplemental irrigation was given until the start of the experiments.

4.2.2 Experimental design

The retention and detention experiments were undertaken from 8 to 30 September 2013. Ventilated rain shelters were constructed over the experimental rain garden modules so that the exact quantities of simulated rainfall could be applied without greatly altering the humidity and evapotranspiration properties. Waterproof luminance diffused polythene clothes were used as the top of the rain shelters to stop the influence of rainfall, while sewn net covers were used to construct the sides of the rain shelters (Fig. 4.6). During the experimental period, the mean inside temperature of experimental modules was 14.4℃, the maximum day temperature was 33.9℃ and minimum temperature at night was 6.1℃.

Fig. 4.6. Experimental rain garden modules with ventilated rain shelters. Photo was taken by the author in September 2013.

4.2.2.1 Retention

The retention tests took place three times under different ADWPs (2 days, 5 days and 7 days) from 8 September to September 23, 2013. The mean daily temperature during the 2-day (22 September to 23 September), 5-day (8 September to 12 September) and 7-day (14 September to 20 September) ADWPs were 17.0℃, 16.5℃ and 11.2℃, respectively. It was noticeable that during the 7-day ADWP, mean daily temperature was significantly less than in the 2-day and 5-day ADWPs. This might contribute to a lower daily evapotranspiration during the 7-day ADWP than the other two ADWPs. In the retention tests, ADWP was the duration that soil has been left to dry from its field capacity to the next round of tests in this experiment. This experiment aimed to observe how the systems respond to ‘significant’ events. For instance, Stovin et al. (2012) considered rainfall events with a return period of greater than one year to be significant. In this study, the designed target retention amount of each module equated to a 1 h rainfall in 10 yr return event that falls onto the 10 m^2 total impermeable drainage area adjacent to the 2 m^2 module surface. The depth of the 10 yr return period 1 h rainfall was given as 21.94 mm based on the full rainfall time series and return period data for Sheffield, which was taken from the FEH CD-ROM (NERC, 1999). The total drainage area (i.e. 10 m^2 in this study) was calculated following the given equations (1) (2) suggested by Woelfle-Erskine and Uncapher (2012):

$$\text{Rain garden area} = \text{runoff volume} \div \text{rain garden ponding depth} \quad (1)$$

$$\text{Runoff volume} = \text{total drainage area} \times \text{rainfall intensity} \times \text{duration of rainfall} \quad (2)$$

The given rain garden area and ponding depth in this study were 2 m^2 and 100 mm, respectively. The equations have been widely used in North American rain garden guides such as “Rain Gardens: A how-to manual for homeowners” (Bannerman & Considine, 2003). It is noticeable that there are various suggested ratios of rain garden infiltration area to the total drainage area. For example, in Australia, FAWB (2009) suggests an extremely low ratio ranging from 2% to 5%, but the rain garden must have a 400-600 mm ponding depth and the soil infiltration rate is recommended to be at least between 100 to 300 mm/h. Emanuel et al. (2010) recommend the size of the rain garden to be at least 10% of the impervious surface draining to the garden, while PADEP (2006) suggests that the ratio of rain garden infiltration area to the total impervious drainage area should generally not exceed 1:5. Therefore, the 1:5 loading ratio (i.e. 2 m^2 infiltration area to 10 m^2 total impermeable drainage area) is considered to be reasonable.

During the retention test, tap water was distributed equally among all fifteen rain garden modules using a mist nozzle to simulate the inputs of this 21.94 mm rainfall (0.2194 m^3 for each module) generated by the adjacent impervious surfaces. A water flow meter (Gardena 8188-20, Husqvarna UK Ltd., Newton Aycliffe, UK) was connected to the mist nozzle to monitor the quantity of water outflow and flow rate. The precision of this water flow meter was within the factory precision of 0.1 L and 0.1 L min^{-1} resolution. The required water input intensity was therefore 0.37 mm/min (3.7 L/min), and this precision implies a maximum error of +/- 3%. The mist nozzle was adjusted to constantly give the simulated rainfall across all rain garden modules at the rate of 3.7 L min^{-1}, and was shut off when exactly 0.2194 m^3 water left the system with approximately 60 minutes. The mist nozzle was kept at a vertical distance of 500 mm from the planting bed and was oscillated to distribute the 21.94 mm simulated rainfall equally to the surface area of each rain garden module as much as possible. The total amount of runoff outflow leaving each module was collected and recorded the next day at 9.00 am after the runoff had stopped. The retention was determined for the difference between the rainfall depth (in mm) and the mean runoff depth (in mm) from the different vegetation types.

4.2.2.2 Detention

The detention tests were carried out separately from the retention experiment on three different days (September 25, 27 and 30, 2013). During the detention tests, a water flow meter (Gardena 8188-20, Husqvarna UK Ltd., Newton Aycliffe, UK) was connected to the mist nozzle to monitor the flow rate. The mist nozzle was adjusted to constantly provide the simulated rainfall at 1.21 mm/min (12.1 L min^{-1}) for an hour across all fifteen rain garden modules. This experiment aimed to look at short duration (e.g. 1 hour event) high intensity events as the worst-case scenarios for detention design in rain gardens. The given depth of water input in the 1 h artificial rainfall in each module was 72.6 mm, which was representative of an extreme event, in excess of 1 in 50 yrs for the FEH design storms depths (Stovin et al., 2012). The mist nozzle was kept at a vertical distance of 500 mm from the planting bed and was oscillated to distribute the artificial rainfall equally to the surface area of each rain garden module as much as possible. All detention tests started with pre-wetted soil media (i.e. irrigate it to its field capacity and let it drain for two hours), to ensure that there was no significant runoff (i.e. the level of runoff that leaked out of the pre-wet system was less than 1 mm in a 10-min period). In this way, all inflow became runoff and retention equalled 0 mm, so that only runoff delay (i.e. the lag time to the peak flow of drainage) and conservation of mass between the rainfall and runoff were checked.

In the detention experiment, the runoff hydrograph from each module was measured for 100 minutes. It was identified that the output flow rate from all treatments reached peak rates within the 100 minutes. At the end of the 100 minutes, runoff was observed to leave the systems with constant flow rate and monitoring ceased due to time constraints. However, without sampling the complete runoff profile, it was not possible to meaningfully calculate the centroid delay (i.e. the time between the centre of mass of the water inflow and that of the runoff outflow). Therefore, runoff delay in each module was determined as the t_{50} delay, which refers to the time between the median value on the cumulative artificial rainfall profile and the same absolute depth on the cumulative runoff profile. The mass of the runoff from each module was measured for 100 minutes at 5-min intervals.

4.2.2.3 Growth of canopy and roots

Maximum plant shoot height (from bottom to highest leaf apex) and spread (i.e. the mean value of the plant length and width when plant shoots were observed from above) of each species were measured. In this study, cover values (the total % cover) of the vegetation among the treatments were estimated visually in the 1×2 m quadrats (i.e. the whole surface area of each rain garden module). On 1 October 2013, representative plants for each species were excavated to the depth of the deepest visible root. The roots were soaked in water for two hours after which the soil was removed from the roots with a fine spray of water, so that the maximum rooting depth (i.e. the deepest depth reached by the plant roots) and the lateral root spread (i.e. maximum one-sided linear distance reached by its roots through the centre of the plant) of the representative plants for each species were measured. In this study, the number of plants for each species in the A series was different due to the nature of its cultivation methodology (sowing seeds in situ), therefore representative plants for each species were chosen randomly from each module. Shoot heights and diameters, as well as the rooting depths and lateral root spreads, were measured to obtain five replicates of each species in total from the A series, respectively. Values of plant shoot and root characteristics for every plant from the B series were measured to determine the mean values. The above parameters were later combined as vegetative communities among the mown grass and mixed forb-rich perennial treatments.

4.2.3 Data analysis

Two-way ANOVA analysis was applied to determine if the runoff retention was significantly affected by the different vegetation types and the antecedent dry weather periods (ADWP), and whether there was an interaction between vegetation types and ADWP. One-way ANOVA was introduced to determine if the different vegetative treatments' detention metrics (e.g. runoff t_{50} delay and peak attenuation), canopy diversity (i.e. the maximum plant shoot height and spread), root diversity (i.e. the maximum rooting depth and lateral root spread), % cover and the number of species were independent of one another. To meet the assumptions necessary for ANOVA, % cover was logit transformed as it was proportion and back transformed for presentation. In ANOVA models for the other tested datasets were checked using Levene's test for normality and homogeneity. No conclusive evidences that the assumptions were infringed and therefore the analyses were continued with. Means were separated by Turkey's test, differences were considered statistically significant for $P < 0.05$. The SPSS 20.0 statistical package was used to perform the above analyses.

4.3 Results

4.3.1 Retention in experimental systems

A two-way ANOVA showed that the different vegetation treatments (P< 0.001) and the different ADWPs (P< 0.001) had significant effects on the runoff retention of the systems. The interaction was significant (P< 0.001). During the retention tests, there were three antecedent dry weather periods (ADWP): 2 days, 5 days and 7 days. The control group with bare soil retained 66.51%, 71.60% and 75.44% of the 21.94 mm artificial rainfall under the 2-day, 5-day and 7-day ADWP, respectively. Mown grasses (A series) retained 69.70%, 70.70% and 71.85% of the 21.94 mm artificial rainfall under the 2-day, 5-day and 7-day ADWP, respectively. Mixed forb-rich perennials (B series) retained 74.59%, 75.93% and 76.73% of the 21.94 mm artificial rainfall under the 2-day, 5-day and 7-day ADWP, respectively.

Fig. 4.7 shows the mean runoff retention among vegetative treatments across the three ADWPs. It is noticeable that the mixed forb-rich perennials consistently had significantly greater mean runoff retention over the two other treatments across all three ADWPs (Fig. 4.7). When the systems only experienced a short ADWP duration (2 days), the control group with bare soils had the least reduction in runoff outlet (Fig. 4.7). During the 5-day ADWP, the reduction in runoff from the control group and the mown grasses were not statistically different from each other, despite that the control group had slightly higher runoff retention than that of the mown grasses (Fig. 4.7). During the 7-day ADWP, the runoff retention in bare soils was significantly higher than in the mown grasses (Fig. 4.7). It could be seen that the two vegetated groups could retain more rainfall than the control group after 2-day ADWP, which means the retention capacity was recovered more quickly in the vegetated treatments than in bare soils. However, the mown grasses only demonstrated higher runoff retention than the control group with bare soils under the 2-day ADWP, while the runoff retention in control group exceeded that of the mown grasses under the 5-day and 7-day ADWPs.

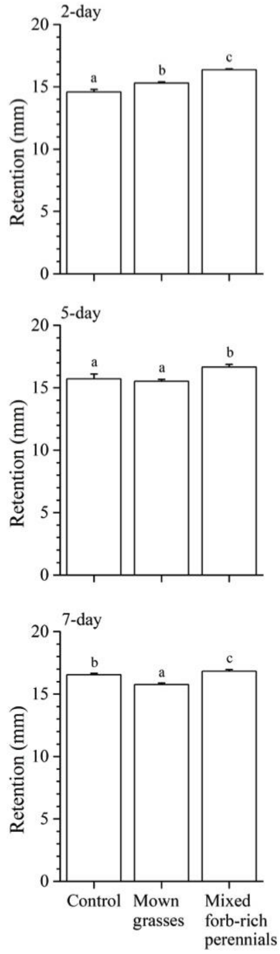


Fig. 4.7. Mean retention of the three different treatments after the 2-Day, 5-Day and 7-Day ADWP (Error bars indicate standard deviation from the mean. Means with the same letter do not differ significantly from each other.)

Increased length of ADWPs had a significantly increased effect on the runoff reduction by increasing retention capacity across all three treatments. Runoff retention from the control group and mown grasses was statistically different among different ADWPs (Fig. 4.8). The control group and mown grasses treatment had the highest runoff retention under the 7-day ADWP, while the least runoff retention was found under the 2-day ADWP (Fig. 4.8). In the mixed forb-rich perennials, runoff retention was found significantly lower under the 2-day ADWP than under the 5-day and 7-day ADWP that were not statistically different from each other (Fig. 4.8).

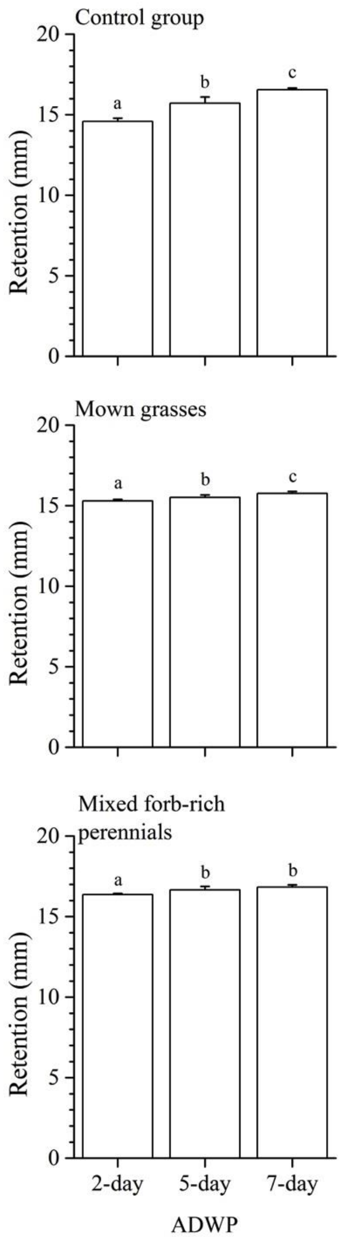


Fig. 4.8. Mean retention after the 2-Day, 5-Day and 7-Day ADWPs in the control group, mown grasses and mixed forb-rich perennials. (Error bars indicate standard deviation from the mean. Means with the same letter do not differ significantly from each other.)

4.3.2 Detention in experimental systems

Fig. 4.9 shows the individual cumulative runoff profiles for the mown grasses (Fig. 4.9a), forb-rich perennials (Fig. 4.9b) and the control group (Fig. 4.9c). During the detention tests, the runoff data collected was very consistent over the 15 trials undertaken over the course of 3 days (September 25, 27 and 30, 2013) in all three groups (Fig. 4.9). The consistent data from all three groups demonstrating a relatively narrow spread during the 100 minutes from the onset of water inputs (Fig. 4.9). This leads to the conclusion that the results themselves are relatively reliable. Due to time constraints of this study, the runoff hydrograph from each module was measured for only 100 minutes. However, the runoff was observed to leave the systems at a constant outflow rate at the end of the 100 minutes. Runoff was then expected to leak out of the systems at the minimum runoff rates within the following hours until it reached the conservation of mass.

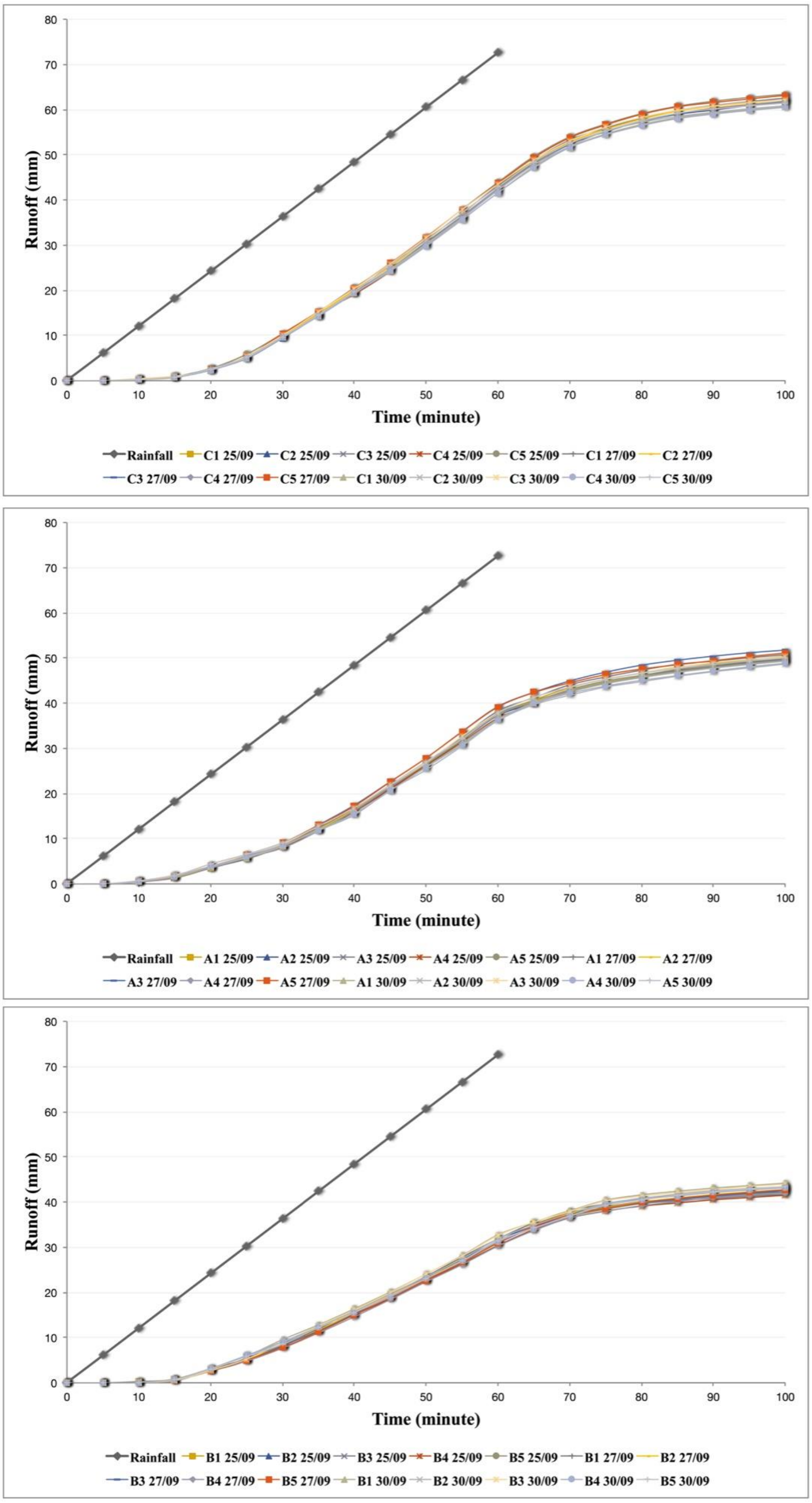


Fig. 4.9. Cumulative runoff profiles for the systems during the detention testing (a: control group; b: mown grasses; c: forb-rich perennials)

Fig. 4.10 shows the runoff rates seen from the detention tests, these were mean values for each 5-minute time step of the five repeats of each treatment. Lower peak runoff rates can be observed from the vegetated treatments compared to the non-vegetated control group (Fig. 4.10). It can be seen that no outflow was leaving the systems of all three treatments within the first five minutes (Fig. 4.10). Runoff rate from the bare soil was lower than in the vegetated treatments within the first 20 minutes. However, it then exceeded the vegetated treatments and reached a peak at 55 minutes from the onset of inflow, before the vegetated treatments reached their peak runoff rate at 60 minutes from the onset of inflow (Fig. 4.10). During the 60 min artificial rainfall event, the peak rate of runoff outflow from the control group reached the water inflow rate, while the vegetated treatments have shown clear attenuation in runoff peak rate compared to the water inflow rate (Fig. 4.10). The runoff peak rate lasted for 5 minutes before falling down in the control group, whereas the runoff peak rate immediately started to drop in the two vegetated groups (Fig. 4.10). After the water inputs were stopped, runoff rates from the mown grasses, the mixed forb-rich perennials and the control group continued to drop over the next 40 minutes to a minimum of 0.15, 0.10, 0.15 mm/min (Fig. 4.10). Mixed forb-rich perennials constantly showed the lowest outflow rate among the three treatments from 35 minutes from the onset of inflow, except between 65 to 70 minutes from the onset of inflow, in which it was slightly higher than from the mown grasses.

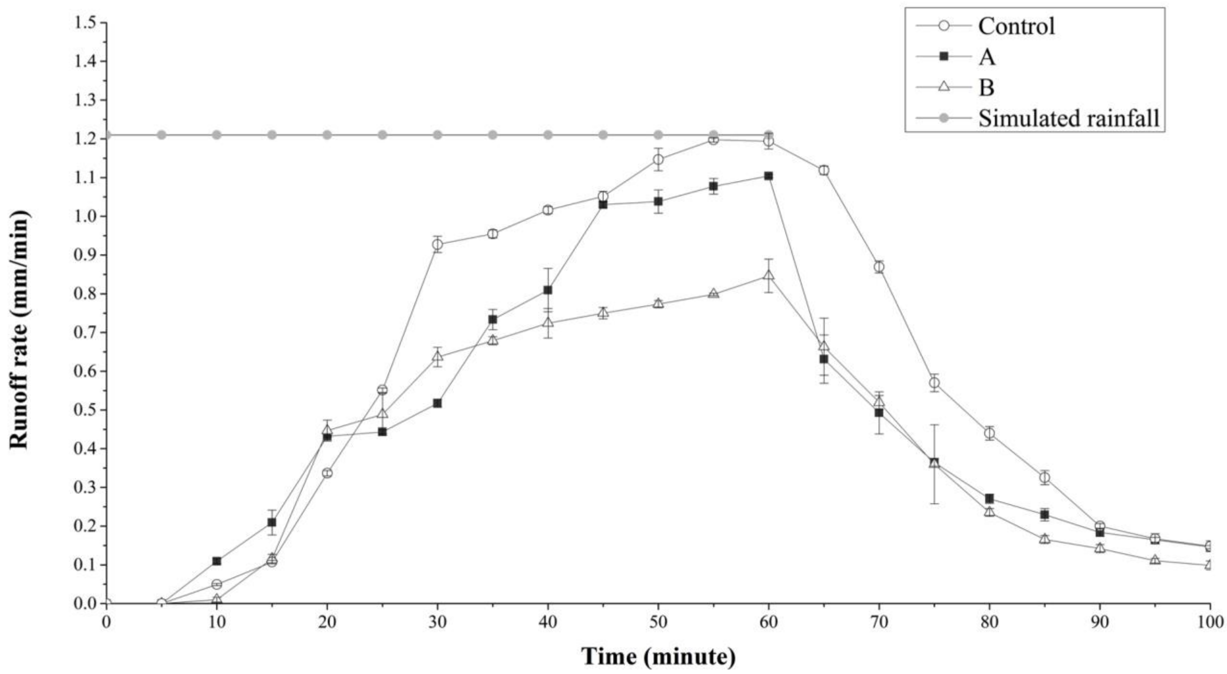


Fig. 4.10. Mean runoff rates over the course of the detention experiment (Error bars indicate standard deviation from the mean.)

Detention effects include runoff t50 delay and the attenuation of peak runoff. Runoff delays were found with the bare soil and the different vegetation types (Fig. 4.11). t50 delay across the control group, mown grasses and mixed forb-rich perennials significantly differed among vegetative treatments (P<0.001), averaging 24.74, 29.02 and 38.18 min, respectively (Fig. 4.11). The mixed forb-rich perennials had the greatest lag time between the first 50% value on the cumulative rainfall profile and the same absolute depth on the cumulative runoff profile, followed by the mown grasses and bare soils (Fig. 4.11).

The runoff peak rates across the control group, mown grasses and mixed forb-rich perennials were significantly differed among vegetative treatments (P<0.001), averaging 1.20, 1.12 and 0.85 mm/min, respectively (Fig. 4.11). The vegetated systems showed a significantly reduced peak runoff rate compared to the non-vegetated control group, where the mixed forb-rich perennials had the best performances on the runoff peak attenuation (i.e. the input rate minus the peak output rate) (Fig. 4.11). The runoff peak rates from the mown grasses and mixed forb-rich perennials were significantly lower than the constant input rate of simulated rainfall (1.21 mm/min) (P<0.001 and P<0.001, respectively). However, a one-way ANOVA showed that the runoff peak rate from the control group was not significantly different from the input rate of simulated rainfall (P=0.127).

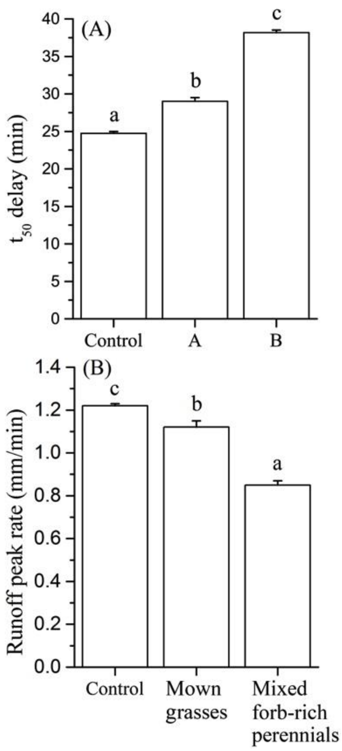


Fig. 4.11. Mean values of runoff delay and peak runoff rates from the three different treatments (Error bars indicate standard deviation from the mean. Means with the same letter do not differ significantly from each other.)

4.3.3 Canopy traits and roots

In the rain garden modules, plant shoot and root growths were determined first by species (Table 4.2) and later combined as vegetative communities for the 1 × 2 m quadrats (i.e. the whole surface area of each rain garden module) to assess differences in canopy traits and roots among the three treatments. Differences in canopy height and spread, rooting depth, lateral root spread, % cover and number of species between vegetative treatments were observed (Table 4.3). The vegetation types differed in shoot height and spread (P<0.001 and P<0.001, respectively), where the mixed forb-rich perennial communities had significantly greater mean values of canopy height and spread over the mown grasses (Table 4.3). Similarly, the means of % cover was significantly higher from the mixed forb-rich perennial treatments compared to the mown grass treatments (P<0.001) (Table 4.3). Rooting depth averaged over all three vegetative treatments was significantly different among each other (P<0.001), whilst the vegetation types differed in lateral root spread (P<0.001) (Table 4.3). The mixed forb-rich perennial communities also had a greater number of species compared to the mown grass systems (P<0.001) (Table 4.3). The results suggested that the mixed forb-rich perennial communities had a higher structural diversity including the canopy diversity, root diversity and species diversity compared to the mown grasses.

Table 4.2. Summary of growth parameters of each species in mown grasses (series A) and mixed forb-rich perennials (series B)

System category	Species	Mean maximum shoot height (cm)	Mean shoot spread (cm)	Mean maximum rooting depth (cm)	Mean lateral root spread (cm)
A	***Agrostis capillaris***	10	5.61	15.34	6.61
	Alopecurus pratensis	10	10.93	9.72	5.12
	Anthoxanthum odoratum	10	12.88	6.81	6.31
	Cynosurus cristatus	10	7.02	6.13	5.04
	Deschampsia cespitosa	10	7.63	8.91	4.91
	Festuca rubra	10	9.6	7.1	6.3
B	***Amsonia tabernaemontana* var. salicifolia**	48.98	13.76	18.91	10.25
	***Astilbe* 'Purple Lance'**	77.72	38.09	19.73	12.79
	Calamagrostis brachytricha	66.32	22.63	23.86	11.54
	Filipendula purpurea	61.73	34.35	26.56	21.87
	***Hemerocallis* 'Golden Chimes'**	69.92	30.11	25.63	19.29
	Iris sibirica	65.61	22.76	26.33	15.09
	Molinia caerulea	82.95	23.36	19.52	12.59
	Rudbeckia fulgida var. deamii	56.38	23.89	19.2	16.06
	Sanguisorba tenuifolia 'Purpurea'	81.86	29.13	21.31	7.78
	Veronicastrum virginicum	98.48	19.35	25.02	10.29

Table 4.3. Summary of vegetation analysis for the 1 × 2 m quadrats

System category	Mean maximum shoot height (cm)	Mean shoot spread (cm)	Mean maximum rooting depth (cm)	Mean lateral root spread (cm)	Mean value of total % cover
A	10.00	8.95	9.00	5.72	76.40
B	71.00	25.74	22.61	13.75	86.80
C	0.00	0.00	0.00	0.00	0.00

4.4 Discussion

In the retention tests of the present study, stormwater input was designed to equal the 10yr-return hourly rainfall. It was shown that different vegetation types and antecedent dry weather period (ADWP) are two key mechanisms governing the hydrologic response (i.e. retention and detention) in rain gardens, which is to be expected. The observed differences are probably a result of the vegetation types altering the antecedent soil water content through the change of evapotranspiration and plant-induced differences in soil structure which may lead to the differences in the potential for rain gardens (Johnston, 2011; Gonzalez-Merchan et al., 2014). Overall, the taxonomically diverse forb-rich perennials were the most effective in the reduction of rainwater runoff, runoff delay and runoff peak attenuation. This indicates the advanced potential of using taxonomically diverse forb-rich plant communities as an alternative vegetation type to replace the conventional vegetation with low species richness and structural diversity (e.g. mown grasses) in rain gardens to contribute to a better stormwater quantitative control performance and help prevent the adjacent urban catchments and drainage system from flash flooding over the traditional mown vegetation.

Plant-induced increase of runoff retention in stormwater management practices were often explained by biomass, sizes and leaf area of adopted plants in previous studies. For example, Nagase and Dunnett (2012) suggested that the greater dry shoot weights of particular species or mixed plantings were positively related to stormwater runoff retention. Their species-specific study also indicated that taller plants with larger spreads tended to retain and intercept more runoff in experimental modules, while shorter plants with smaller diameters tended to shed more runoff (Nagase & Dunnett, 2012). Waring and Landsberg (2011) and Vanuytrecht et al. (2014) suggested that greater leaf mass and leaf area are effective in increasing evapotranspiration rate so that a greater loss of runoff from stormwater retentions could be governed by vegetation with greater canopy growths. Although the biomass of different vegetation types were not formally accessed in this study, the greatest plant shoot heights/spreads, rooting depths and root spreads, as well as the % cover found in the mixed forb-rich perennials helped explain their advanced retention performances over the mown grasses and bare soils.

We also assume that the greater rooting depths and lateral root spreads in the mixed forb-rich perennials contributed to greater retention compared with the mown grasses. Similar results were concluded that deep-rooted (Barrett et al., 2013) and greater rooting volume (Passeport et al., 2009) could contribute to a higher water retention in stormwater management facilities. Deeper roots and greater root expansion means plants may extract greater volume of excess water in soils and transpire it through leaves back to the atmosphere (Razzaghmanesh & Beecham, 2014), which thus recover the retention capacity of absorbent substrates. Root growths may increase and enlarge pore spaces in soil to allow greater absorption of stormwater in soils (Archer et al., 2002), which may also maintain the soil hydraulic conductivity over time to sustain infiltration capacity (Lewis et al., 2008). However, it is also noticeable that the basins of the experimental systems in this study were sealed with impervious liners, which were therefore assumed to have no infiltration. It means the preferential flows through the soils from the systems were largely turned to runoff. The retention observed in this study were thus mainly resulted from soil absorption, evapotranspiration and plant uptake. Stormwater retention in practical rain garden allowing natural infiltration is thus expected to be more effective than the currently reported experimental systems.

In this study, the mixed perennials which were richer in species showed greater runoff retention than the mown grasses adopted less number of species. We therefore assumed that the promoted primary function of rain garden (i.e. increase in runoff retention) is not simply a result of using species with greater biomass, sizes and leaf area, but may also be promoted by increasing the vegetation species richness. This argument has been supported by evidences of previous studies. For instance, Dunnett et al. (2008) concluded that a more diverse species composition could retain more water input than the communities adopted less species. Johnston (2011) suggested an up to 10-fold increase in soil saturated hydraulic conductivity was observed in practical rain gardens planted with the mix of a number of prairie grasses and forbs compared to the models adopted a much less diverse turfgrass mix.

Rixen and Mulder (2005) suggested that greater species richness allows a wide variety of canopy traits in different species to build greater structural diversity, which may result in a greater rainfall interception governing the parallel increase in runoff retention and species richness. Taxonomically diverse vegetation allows different species with a wide variety of root architectures to have dynamic root penetrations to reach different part of soil, and then extract more soil moisture for loss through leaf transpiration. Allen et al. (1998) concluded that air movement is vital to evapotranspiration. Taxonomically diverse plantings assembled with species of different heights may have more porous spaces in their aboveground traits to allow better air movement to reduce moisture concentration in leaf area and therefore encourage the total loss of runoff through evapotranspiration. In contrast, grass swards mown to same height may hamper the air movement, which may help explain the less and limited runoff loss through this vegetation type in this study compared to the mixed forb-rich perennials. Furthermore, if tall plants are surrounded by contrasting shorter species, the ‘clothesline’ effect on canopy conductance on a warm, windy day can result in proportionally high water loss through evapotranspiration (Allen et al., 1998), which is expected to strengthen the evapotranspiration in the structural diverse perennial in this study that mixed with species of different heights.

Whilst there is some evidence in the literature that increased species diversity may contribute to enhanced retention over and above the effects due to plant biomass alone, the experimental design does not permit the effects of biomass and species diversity to be separated here. This aspect is clearly something that would be worthy of further research.

In this study, parallels between the retention capacity and the given length of ADWP were found, as has been suggested by Stovin et al (2012) and Mangangka et al. (2015) as a result of the reduction of antecedent soil water content to restore the system's retention capacity via evapotranspiration and discharge. The interaction between vegetation types and ADWP also significantly altered the drainage dynamics in the experimental rain garden modules.

In Fig. 4.7, the systems' retention in treatments planted with mown grasses and mixed forb-rich perennials were 0.70 mm and 1.77 mm higher than the bare soil after a short period of ADWP (2 days), which equal to 4.80% and 12.14% of the retention caused by soil (14.59 mm) respectively. In the present study, we assume this part of retention was caused by transpiration from vegetation and plant uptake. However, decreasing trend in runoff retention resulting from the transpiration through plant foliage and plant uptake in vegetative treatments was observed across increased ADWPs. Retention resulting from vegetation dramatically fell to -0.20 mm and 0.95 mm in mown grasses and mixed forb-rich perennials after 5-day ADWP, respectively, while retention resulting from soil absorption and evaporation reached 15.71 mm (Fig. 4.7). After 7-day ADWP, retention caused by soil greatly increased to 16.55 mm, whereas the retention resulting from vegetation took a further decrease to -0.79 mm and 0.28 mm from the treatments of mown grasses and mixed forb-rich perennials, respectively (Fig. 4.7). The stated evidences showed that after a longer given duration of ADWP (5 days and 7 days), the effectiveness of rain garden retention was dominantly governed by soil absorption and evaporation.

It is also noticeable that only mixed forb-rich perennials positively contributed to water loss throughout the whole retention test, whereas negative impact caused by mown grasses on the retention capacity restoration through the increasing length of ADWPs was found. Allen et al. (1998) suggest that air movement would be slowed down due to surface friction, which is slowest at the soil surface and increases with height. It is possible that the low growing mown grasses sheltered the soil, hampering water loss through evapotranspiration in this study. It again demonstrated the advantages of taxonomically diverse plant communities as an alternative vegetation approach for rain gardens. Although there is incremental differences in the two retention levels of mixed forb-rich perennials due to 5- and 7-day ADWPs, it is noticeable that the two data were not statistically significantly different from each other. It might be caused by the lower temperature during the 7-day ADWP (mean daily temperature 11.2℃) than that of the 5-day ADWP (mean daily temperature

16.5°C), where the significantly increased temperature during the 5-day ADWP could potentially increase the evapotranspiration to increase the reduction of runoff than during the 7-day ADWP.

In most instances, storm water will immediately turn to runoff when it falls onto the impermeable pavements in an urban area, thus there is no delay of runoff peak. The delay of runoff is because water inflow must filter through the soil, filter and drainage layers before it can outflow from the system. In this study, differences in runoff delay between the vegetated treatments and the non-vegetated soil treatment were determined. A significantly longer t_{50} delay was found from the mixed forb-rich perennials than from the other two treatments, where the bare soils had the least lag time. Opposite results were reached in previous studies as the experimental observations were restricted by test conditions. For example, Johnston (2011) reported that the bare soil had greater lag time to peak flow than the turfgrass and prairie treatments that were not different from one another. This was because of Johnston (2011) did not test the different vegetation types in pre-wetted soil conditions (i.e. detention was not separated from retention effect). Greater runoff delay is expected in vegetative treatments than in bare soils, as vegetation canopies will intercept a portion of rainfall as well as reducing the velocity of runoff to allow more time for infiltration. Plant root system can elongate soil micropores following root turnover (McCallum *et al,* 2004; Ball *et al,* 2005), reverse soil compaction and enhance soil porosity (Yunusa & Newton, 2003) so that more water would be temporarily stored in the increased macrospores, and therefore runoff moves through the vegetated systems slower. Different plant traits are therefore expected to influence runoff lag time differently. The significantly greater canopy % cover, canopy diversity, rooting depth and root expansion in the mixed forb-rich communities compared to the mown grasses help explain the better detention behaviour in rain garden models that planted with forb-rich perennials over that of mown grasses. It is also worth noting that the impervious basins of the experimental systems would eliminate infiltration and reduce the runoff delay. Thus, the practical field rain gardens would be expected to have a longer runoff lag time than the stated results in this study.

Another benefit of the rain garden is often reducing the peak outflow of the system compared to if no such facility had been implemented. In this study, the results from the detention experiment indicated that the vegetated systems had significantly promoted the runoff peak attenuation, while the non-vegetated control group with only soil amendment showed no significant contribution to the runoff peak attenuation. The results again showed the importance of the adaptation of vegetation in rain gardens for the detention of storm events. Differences in vegetation type in the experimental rain garden modules markedly altered the runoff peak attenuation in that the taxonomically diverse forb-rich perennials had the greatest attenuation in the runoff peak, followed by mown grasses and bare soil.

4.5 Conclusion

This study has shown that the vegetative choices could influence the hydrologic behaviour of rain gardens. The length of ADWPs was determined to significantly affect retention. Although the vegetated treatments showed limited increase of retention capacity restoration compared to bare soil when the given days of ADWPs increased, the mixed forb-rich perennials were the most effective in reducing water runoff over all given ADWPs (2 days, 5 days and 7 days), while the mown grasses retained the least amounts of runoff in most of the instances over the 5-day and 7-day ADWP. The taxonomically diverse forb-rich perennials with higher plant diversity and structural diversity were the most effective for runoff delay and attenuating runoff peak rate, followed by mown grasses and bare soil. All the results conducted in this study argued that the most common conventional vegetation with low species richness and structural diversity (e.g. mown grasses) which is grown in the contemporary rain gardens and other stormwater management facilities are very limited in improving either stormwater retention or detention.

Urban rain gardens to date are often engineered without enough consideration of horticulture and ecology, which thus lead to limited primary functions and functional diversity. Experimental outcomes in this study suggest that the use of suitable plants that have larger spreads and greater heights, as well as plants with deep roots and greater root expansion could govern the hydrologic performance of rain gardens.

Results of this study also indicates that vegetation which is richer in species than those adopted mown turfs benefits the on-site stormwater quantitative control. Monoculture and mixtures with low species richness also has been long-time criticised in practices for their limitations with respect to biodiversity and ecosystem services (Vanuytrecht et al., 2014; Van Mechelen et al., 2015). Therefore, the future design for urban rain garden vegetation should take a step forward to embrace species diversity. We therefore recommend the future planting design to adopt as many suitable species as possible in rain garden vegetation, so that the different heights, various canopy properties, different foliage shapes and complicated bio-root zones of different species may maintain the complexity and integrity of its structure and functional diversity over time.

Furthermore, parameters such as leaf mass and higher leaf area index (LAI), root biomass, soil media depth and composition, stormwater input size and the phenology of the vegetation that affected the runoff distribution in rain gardens were not explicitly studied. All these factors are suggested to be involved in the future research to obtain a better understanding of the plant-induced effects in stormwater management. This will increase the utility and accuracy of models predicting the hydrologic performances of rain gardens.

Due to the time constraint in the present study, the mixed forb-rich perennials were implemented by planting rather than by sowing seeds. Sowing seeds in-situ as the alternative vegetation method would promote a greater density of vegetation.

Establishing vegetation by sowing may produce a more naturalistic random pattern and much more complicated aboveground traits and root systems compared with the mown vegetation and border-like plantings developed by planting (Dunnett & Clayden, 2007). In future research, the mixed forb-rich plantings intended for use in rain gardens are suggested to be implemented by sowing in-situ to evaluate the effectiveness of this vegetation.

In addition, experiments in this study only conducted at one season of year. However, phenological development of the vegetation may alter the hydraulic and hydrologic performance of rain gardens (Waring & Landsberg, 2011). Knowing the phenology effect on the retention and detention of rain gardens adopted different vegetation types may provide more insights to predict the hydrologic dynamics of rain gardens and to provide corresponding planting suggestions, and is thus suggested to be included in future study.

Chapter 5: Evaluation of the effects of simple weed control means and dynamic moisture distribution on sown forb-rich plantings in rain gardens under the UK weather

5.1 Introduction

5.1.1 Vegetation techniques in typical rain gardens

As noted previously in Chapter 2, recent efforts in rain garden research have mainly focused on the development of alternative design configurations and amending soil media compositions (Barrett et al., 2013; Dietz, 2007; Hsieh & Davis, 2005), whereas there is limited research into the vegetation techniques for use in rain gardens. Contemporary vegetation techniques in urban landscapes, and especially sustainable stormwater management systems, have been consistently criticised for their monotonous appearance and rather limited species diversity, low biodiversity, as well as the excessive involvement of resource-consumptive horticulture technologies which may even be harmful to the urban environment, such as the wide use of herbicides for weed control (Grime, 2001; Yang et al, 2013).

In the UK, vegetation mixes that are seen in most rain gardens and many other sustainable stormwater management components might range from those with extremely low species richness which may only provide minimal habitat value and poor aesthetic values (see Chapter 4, Fig. 5.1), through to spontaneous communities of competitive tall forbs and grasses which are rather limited in aesthetic value (Fig. 5.1). Similar examples can be found in local authority or city government spaces in the US (Fig. 5.2). Many rain garden applications with border type plantings can be found in domestic gardens or community spaces in the US (Fig. 5.3), and these can be more ornamental (Steiner & Domm, 2012). The border plantings are valued in rain gardens because of their dramatic flowering displays. However, these features may significantly raise the financial, labour and resource inputs as they largely employ transplanting mature plants or seedlings as the main vegetation technique. Such techniques could be rather expensive over large areas, and might leave large gaps of bare soil between plants in early stages that not only provide unpleasant landscape but also may provide spaces for weed colonisation (Dunnett & Clayden, 2007).

Fig. 5.1. Bioswale featured a spontaneous plant community with poor visual appearance, Sheffield, UK. Photo was taken by the author in July 2013.

Fig. 5.2. Rain gardens installed alongside city road, which featured monoculture of native grasses, Columbus, USA. Photo was taken by the author in October 2012.

Fig. 5.3. Private rain gardens with more ornamental border plantings, USA, Source, Steiner & Domm, 2012.

Moreover, the traditional methods of management in urban plantings may be applied in rain gardens, which can require excessive maintenance such as wasteful seasonal mowing, changing plants for various flowering appeal in different seasons and additional weeding (Hitchmough & Dunnett, 2003). Mowing may not only compromise the visual appeal of plantings but may also turn rain gardens into a muddy quagmire in the wet season, thus expense the features' efficiency in runoff control (Steiner & Domm, 2012) and allowing pollutants to be released upon plant decomposition (Schultz, 1998). Changing plantings seasonally is not common practice in rain gardens, but it is understandable that people may replace the existing plants for different seasonal flowery scenes. Plant diversity is therefore particularly important to sustain the display of plantings in rain gardens, which is also vital to the conservation of local biodiversity (Dunnett & Clayden, 2007; Steiner & Domm, 2012). Weeding is essential for sustaining the establishment of the desired plant communities, however, intensely weeding in weedy urban soils would require additional budget and labour. Using herbicides is the most widely used and effective way of weeding (Hitchmough et al., 2008), however, the use of herbicides in rain gardens is extremely harmful, as these features are often connected with public drainage that may discharge the polluted excess water into local water bodies and cause concentration of herbicides to damage the aquatic ecosystems (Yang et al., 2013).

There is, therefore, a real need to look for opportunities to have creative and low-cost vegetation in rain gardens, and particularly those that can be established easily onsite and managed by extensive techniques, as well as extending plant diversity, ecological value and aesthetic values.

5.1.2 Sowing seeds in-situ as an alternative vegetation method in typical rain gardens

Sowing seeds in-situ is a well-established method for creating taxonomically diverse and structurally complex plant communities (e.g. wildflower meadows and prairies) (Hitchmough et al., 2008), which has been adapted into urban areas from the early 1980s in Britain (McDonald, 1993). Seeding as an alternative vegetation establishment technique to the traditional planting methodology, has gained popularity because of its cost-efficiency over large areas with minimal labour input (Dunnett & Clayden, 2007). Sowing in-situ is also particularly useful for establishing vegetation that mimics the naturalistic display and adding biodiversity value, as well as requiring extensive maintenance such as minimal mowing (Hitchmough & Wagner, 2013). Generally, sowing in-situ has the following advantages in response to the requirement of low-input and low-impact design of urban vegetation:

(1) Potentially producing a fully naturalistic visual appearance in vegetation, creating taxonomically diverse plant communities (Hitchmough & Dunnett, 2003),

(2) It is cost-efficient to cover large areas with vegetation (Dunnett & Clayden, 2007),

(3) Reducing the energy and irrigation consumption compared with generating plug and pot plants in a greenhouse,

(4) Results in a greater density of desirable plants per square metre, thus reducing the weeding input (Dunnett & Clayden, 2007).

The use of sown taxonomically diverse communities in the urban context is inspired by natural habitats. It has a variety of possible ecological and social merits such as the urban nature conservation and interaction with local biodiversity, as well as contributing to the amenity value of the local environment (Batáry et al., 2012; Hitchmough & Dunnett, 2003; Hitchmough et al., 2008). There are good examples of sown taxonomically diverse communities as biodiversity conservation practices in urban areas. For example, Losvik and Austad (2002) grew a sown species-rich meadow in western Norway, which successfully established 16 endangered species in local hay meadows. In this study, the traditional meadow management means, including the application of artificial fertilisers, intensive grazing (e.g. 1-2 weeks) in spring, autumn, as well as the late cut in August, were replaced by creating 16 cm gaps for each 0.5 m in the meadow field and only three extensive cuttings in June, August and October (Losvik & Austad, 2002). The extensive maintenance resulted in a greater species-richness in the meadows, while the numbers of the endangered species occurring in the field were increased (Losvik & Austad, 2002). Similar results were suggested in a number of studies that plant diversity and the number of preferred endangered species in meadows were significantly decreased due to traditional management regimes such as a combination of intensive mowing and grazing (Losvik, 1996; Losvik 1999) and heavy fertilising (Tilman 1993).

Increasing the sown taxonomically diverse communities in urban areas for their various ecosystem services and visual amenities can use either the native seed mixtures obtained from the remaining species-rich local meadows (McDonald, 1993; McDonald et al., 1996), or the seed mixes purchased from seed suppliers. For instance, McDonald (1993) sowed seed mixtures of water meadows harvested from the species-rich wild meadows in Oxey Mead on the outskirts of Oxford, UK, onto the banks of the river Thames in Somerford Mead, which is 2 km upstream from Oxey Mead. In this study, seed mixtures were collected in July 1985 and seeded in October 1986, with only one cutting for hay at the end of June and subsequent sheep-grazing, the water meadow communities in Somerford Mead were successfully established, with 49% constant species were reported as the desirable species (McDonald, 1993). Commercial meadow mixes are widely applied in urban landscape, which are reported as typically consisting of approximately 80% grass species and only 20% forb species (Wells et al., 1989), while the species composition are either expressed as the percentage of seed number or the percentage weight of seeds. Nonetheless, grass species in the commercial mixtures could be competitive to the forb species even though they are less competitive than the most productive weeds recruited from the soil seed bank in urban areas (Mitchley et al., 1996). The typical commercial meadow mixtures are not forb-rich, which means they are not rich in flowers and colours and are therefore limited in aesthetic values. To date, landscape architects are interested in developing the more ornamental plant communities grown from seed mixtures that are deliberately assembled with a diversity of flowery forb species and fewer grass species for a long and dramatic flowering display (Hitchmough & Dunnett, 2003; Hitchmough, 2011). For example, the "Landlife" agency in the UK has been creating various exciting flowering urban meadows from the composition of sown native wildflower seeds (Luscombe & Scott, 2004). Such sown forb-rich communities are developed as

"Creative Conservation" which not only considers the objective ecological merits, but also reflects the dominance of social context (Luscombe & Scott, 2004).

Seeding is a recognised technique for vegetation establishment in margins of wetlands or on the slope and banks of rivers, streams and other water bodies (McDonald, 1993; McDonald, 2001), where the soils are typically moist, similar to rain gardens. To date, people started to sow seeds in-situ as a valuable planting method in rain gardens and other sustainable stormwater management components such as bioswales (Dunnett & Clayden, 2007), and there are examples such as seeding perennial turf mixtures (Mazer et al., 2001), sowing monocultures of perennial ryegrass (Katuwal et al., 2008) and seeding shrub species (Skabelund, 2008). However, sown forb-rich communities with great plant diversity in flowering herbaceous species are rarely seen in rain gardens and other sustainable stormwater management components with similar shallow depression structures (e.g. bioswales). A good example can be found in the North Park of the 2012 Olympic Park, London, UK, where the bioswales designed by Nigel Dunnett were created by sowing forb-rich seed mixes (Fig. 5.4). In these bioswales, diverse communities of selected British native species which can survive periodic inundation and dry conditions such as *Leucanthemum vulgare* and *Lythrum salicaria* were established from sown seed mixes (Oudolf & Kingsbury, 2013). These sown native forb-rich communities use the stormwater captured in the swales for irrigation and to facilitate attractive flowering displays. Hitchmough and Wagner (2013) carried out a 5-year investigation of rain garden type diverse sown communities. In spring 2004, a designed rosette-forming forb community, which included 12 non-native and 6 native species hailed from wet grassland habitats and dominated with *Primula* species, was sown into wet, seasonally anaerobic soils to investigate their performance (Hitchmough & Wagner, 2013). There were 12 experimental plots, six of which were irrigated weekly with 8 mm of artificial rainfall from May to August, while the rest of the plots were non-irrigated (Hitchmough & Wagner, 2013). In this study, the most obvious finding was the sown *Primula* total and individual biomass was increased with increased summer soil wetness, which was also associated with enhanced flowering display (Hitchmough & Wagner, 2013). Hitchmough and Wagner (2013) also suggested that once these relatively unproductive rosette-forming forbs were established at high densities, the more productive invasive species could hardly colonise into the sown communities.

Fig. 5.4. Bioswales created by sowing forb-rich mixtures in the North Park of the 2012 Olympic Park, London, UK. Photo was taken by the author in August 2013.

5.1.3 Weed control in sown forb-rich plantings

The use of seeding is one approach that is cost-effective for producing very naturalistic results, as well as adding ecological benefits. However, there are major problems in urban contexts in seeding rain gardens, mainly because urban soils tend to be productive and weedy (Hitchmough et al., 2008). Ruderal species recruited from the soil-stored seed bank in urban areas often have higher levels of seedling establishment and are often more competitive and productive compared to the relatively unproductive ornamental forbs, which may out-perform the desirable sown forbs (Prentis & Norton, 1992; Hitchmough et al., 2008; Sluis, 2002; Gao et al., 2013). Moreover, many forb species that hail from wetter environments tend to be relatively aggressive and productive (Dunnett & Clayden, 2007) and therefore weed competition in rain gardens may be severe.

Suppressing the weedy species might be achieved by the use of herbicides in newly sown plantings (Hitichmough et al, 1994; Hitchmough et al., 2008), the application of regular cutting to the faster growing ruderal species across growing season to mitigate their elimination to the slow-growing desired species (Morgan, 1997; Kleijn, 2003), increasing sowing rates (Hitchmough et al., 2008) and the removal of productive weed-rich topsoil (Hitchmough et al., 2004). Hitchmough et al. (2008) carried out a 3-year (2000-2002) investigation in a sown forb-rich community consisting of 10 British native species and 10 continental European-Eurasian species that were collected from seasonally dry, relatively infertile and calcareous soils. In this study, weeding treatments in the sown communities were designed to make paired comparisons, which included: applying graminicide and without using graminicide, and the use of lower and higher sowing rates (30 and 60 seeds per species for each trial bed, respectively) (Hitchmough et al., 2008). One cutting was applied in June and August 2000, respectively, to keep the communities at a height of 50 mm, and a late cut-off near ground level was applied in September 2000 (Hitchmough et al., 2008). In 2001 and 2002, one cutting was applied in March to maintain the communities at a height of 50 mm, which were then cut near ground level in August (Hitchmough et al., 2008). The experimental observations suggested that the use of herbicides is a more immediate means in suppressing productive weeds, which also significantly increases the density and biomass of the desired sown forbs during the experimental period (Hitchmough et al., 2008). Hitchmough et al. (2008) suggested that the application of the higher sowing rate contributed to a higher density of the sown forbs in community during the whole experimental period of three years, and increased the species richness of the sown forbs in the first two experimental years, but was determined to have no significant effect on increasing the biomass of the sown forbs. The interval cutting back to 50 mm during the first year had no contribution to the mitigation of grass competition (Hitchmough et al., 2008).

In practice, the combination of chemical and physical weed treatments is widely applied. For instance, in the trials of sown rosette-forming forb communities in simulated rain garden soil conditions carried out by Hitchmough and Wagner (2013), the former weeds onsite were eliminated through the application of a glyphosate herbicide. In this study, the dead turf and topsoil was removed to a depth of 75 mm, and a 75 mm weed free mulch mixed with 50% compost and 50% deep-subsoil was laid on top of the ground to prevent weed seed emergence from underlying soils (Hitchmough & Wagner, 2013). Yearly application of herbicide is also recommended in the established sown plantings for minimising weed competition (Hitchmough et al., 2008).

However, weed control management is often problematic in practice (Hitchmough et al., 2008). For instance, as noted previously, the use of herbicides is problematical as they may be particularly harmful to downstream aquatic ecosystems, especially when applied in rain gardens. Suppression of weed emergence from the seed bank in urban soils such as the complete removal of topsoil is costly but may cause severe damage to the productivity in urban soil (Swash & Belding, 1999). There are less interventionist approaches reducing the competition from spontaneous weedy plants in sown plantings. For instance, mulching sterile or weed-free substrates into which forb-rich seed mixture is sown may reduce the potential dominants and competitors' competitive capacity (Mahound & Grime, 1976; Hitchmough et al., 2005). Dunnett and Nolan (2004), Getter and Rowe (2006) concluded that the mulch depth is an important factor influencing the growth of herbaceous plants, and that increasing mulch depth can contribute to a significant improvement in plant growth. Greater mulch depth is therefore assumed to potentially help the sown species to establish early advantages for weed competition. Deep ploughing that inverts topsoil beneath subsoil may also reduce the weed emergence (Luscombe & Scott, 2004). There are some environmentally-friendly experimental techniques, which are potentially good for weed control in sown plantings but, nonetheless, rarely used. For example, the horticultural specialist agency Lindum Turf (York, UK) suggests the application of a felt mat made from recycled cotton and wools to lay over the top of existing ground as a barrier to suppress the emergence of unexpected weedy species but allow the roots of sown species growing through. However, the main concern is there has been little data-based research that has demonstrated the effectiveness of these less interventionist weed control means on the sown forb-rich seed mixture in typical rain gardens conditions. Therefore, there is a real need to investigate simple weed control techniques that enable successful vegetation establishment in rain gardens.

5.1.4 The effects of moisture distribution in rain gardens on sown communities

The typical dynamic soil moisture distribution in rain gardens may be another factor to affect the success of the establishment and development of forb-rich sown plantings. As noted previously, rain gardens depend on seasonal precipitation as their water source, thus a cyclic flooding consists of a cycle of periodic flooding and draining, which may be repeated over time (Dylewski et al., 2011). Due to the effects of cyclic flooding conditions, the habitats in rain gardens are similar to seasonal wetlands that may transform from marshlands to dry lands. Rain gardens have the typical structure of shallow depressions, where gravity and the depression structure may result in a dry-wet gradient of soil moisture distribution throughout the vertical layers of three saturation zones (i.e. margin, slope and bottom). Therefore, typical rain gardens may have a long-term waterlogged bottom, moderate moist slope and dry upland margin (Dunnett and Clayden, 2007).

However, relatively little is known about the effect of soil moisture distribution in rain gardens or other sustainable stormwater management components with similar shallow depression structures (e.g. bioswales) on sown communities. Mazer et al. (2001) established sown grass mixes in three bioswales in northern King County, USA, which were planted with four native perennial turf grasses including *Agrostis stolonifera*, *Festuca arundinacea*, *Poa pratensis*, *Alopecurus geniculatus* and *Festuca ovina*. The topsoil in these bioswales was replaced with sandy loams to a depth of 15 cm, in which the seed mixes were sown at a rate of 4 kg per 100 m^2 (Mazer et al., 2001). The three field bioswales were irrigated by natural precipitation, while the 2-year (1997-1998) experimental observations suggested that the persistent multi-day inundation in bioswales had severely limited the germination and growth of the selected grasses (Mazer et al., 2001). The tolerance of the four selected turfgrass species to cyclic inundation (2-4 cm over substrate surface) were also tested in controlled greenhouse conditions, where seeds were sown into small pots and were treated for two 14-day cyclic flooding: (a) moderate moist soils with controlled irrigation, (b) a dry condition with a 2-day inundation and 12 days without irrigation, (c) an intermediate moist condition with 7 days inundated and 7 days without giving water, and (d) a wet condition with a 12-day inundation and 2-days of withheld irrigation (Mazer et al., 2001). Mazer et al. (2001) presented similar results from the greenhouse survey that persistent inundation resulted in significant suppression in seed germination and seedling growth, where the wet conditions with the longest inundation and shortest draining showed minimal germination and seedling growth amongst all species, and all species grow best when pots were free from flooding. However, as noted previously, Hitchmough and Wagner (2013) provided contradictory results that increased soil moisture in summer significantly increased the biomass of total vegetation and individual species. The major concern in the study of Hitchmough and Wagner (2013) is that the experiment was set up in an open field that became rather dry in summer other than to be applied to practical rain gardens. Therefore, applying extra water had a beneficial effect on plants such as the *Primula* species that do not prefer dry conditions. It is understandable that the outcomes in practice are often restricted by site-specific conditions and the tolerance of selected species, however, the main concern of both studies is that the sown vegetation and individual species' response to the 'dry-wet' moisture gradient and their distribution throughout the typical 'margin-slope-bottom' profile in rain gardens or bioswales are unreported.

Typical cyclic flooding conditions and the hydrological regimes along the ‘dry-wet’ gradient in rain gardens may affect the germination and establishment of sown plantings in several ways. Firstly, runoff may flush seeds away or cause erosion to affect the establishment of sown communities. Altering wet and dry conditions could affect plant establishment of individual species and community from the seed bank by desiccating moisture-needy plants or inundating terrestrial plants from arid habitats. Inundation may stimulate or inhibit the germination of certain species (Brock & Britton, 1995; van der Valk, 1981). In a natural wetland habitat, intermediate frequency of floods creates establishment opportunities for moisture-preferred plant species and prevents competitive exclusion, while dry phases provide opportunities for plant species hailing from arid habitat to establish (Bornette & Amoros, 1996). At the same time, modifying the hydrological regime in the soil may affect the competitive interactions among species for moisture (Keddy & Reznicek, 1986), and change the soil oxygen availability to affect species distribution due to species tolerance of anoxia (van den Brink et al., 1995).

However, the lack of data-based research looking into the interaction between planting establishment in a sown forb-rich planting and the dynamic soil moisture distribution throughout the ‘margin-slope-bottom’ saturation zones in typical rain gardens not only leaves a great research gap but also results in uncertainty regarding the effectiveness of sowing seeds in-situ as the vegetation technology in rain gardens.

5.1.5 Research objectives

The main aim of this work is therefore to look at simple and effective weed control techniques for in-situ sown forb-rich planting in typical rain gardens. In the consideration of cost-efficiency and low-impact to the local environment, this study is particularly interested in two potential techniques: the use of weed free mulches with different mulching depths, and the use of the felt moisture mat provided from Lindum Turf (York, UK). It is also valuable to evaluate the interaction of sown forb-rich plantings and the dynamic moisture distribution throughout the 'margin-slope-bottom' saturation zones in rain garden's depression structure. Therefore, knowing the performance in the establishment of sown communities and individual species throughout the 'dry-wet' saturation zones in a typical rain garden is another focus of this study.

The specific objectives of the present study included:

(1) To investigate the use of mulch and felt mat as two key weed control techniques for an in-situ sown forb-rich seed mix in a series of field rain gardens. It will be achieved by monitoring the emergence of seed mix and the % coverage of vegetation in different treatments,

(2) To determine the effects of the gradient of moisture availability throughout the 'margin-slope-bottom' rain garden profile on the sown forb-rich mix,

achieved by monitoring the emergence of seed mixes (e.g. germination and % coverage) in the ‘dry-wet’ saturation zones.

5.2 Methods

An experiment was established to investigate mulching (two depths including 20 mm and 100 mm) and felt mat (with or without) as the two key weed control techniques for sown forb-rich plantings in rain gardens, and their influence on the emergence and % coverage of sown species through the ‘dry-wet’ moisture gradient, at the premises of Green Estate Ltd. (1˚26’11’‘W, 53˚22’37’‘N), Sheffield, United Kingdom.

5.2.1 Site description

The greater region of Sheffield is generally characterised by a northern temperate climate. Between 1971 and 2000, Sheffield had a mean yearly maximum temperature of 12 ℃ with July being the warmest month, and a mean minimum temperature in January and February of 1.6 ℃, and a normal annual precipitation of 824.7 mm (Stovin et al., 2012).

The study site was fenced to mitigate human disturbance. The trial area for establishing the experimental rain gardens was a narrow area with a size of 38.8 m (length) × 3.4 m (width) along the edge of a field, backed by a hedgerow (Fig. 5.5).

The hedgerow on the north of the trail area was planted as a boundary between the study site and a local primary school (Fig. 5.5), and was pruned to maintain a height of 1.5 m during the whole study. The site sloped to approximately 1:50 (1 vertical to 50 horizontal) sloping to northeast. The aim of the rain gardens was for dewatering the surface runoff generated on the ground of the orchard field and receiving stormwater runoff from outside the site. Excess water from the rain garden was discharged to a retention pond set at the northeast end of the existing site (Fig. 5.6).

Fig. 5.5. The existing trial area. Photo was taken by the author in May 2012.

Fig. 5.6. The retention pond on the northeast end of the study site. Photo was taken by the author in October 2013.

Initially, the existing trial area was uncultivated. Prior to the preparation for this study, the whole site (including the trial area and the orchard field beside the trial area) was very weedy (Fig. 5.5) and supported early successional grassland dominated by *Atriplex patula, Arrenatherum elatius, Chenopodium album, Chamerion angustifolium*, *Galium aparine, Plantago lanceolata*, *Polygonum persicaria, Ranunculus repens*, *Rumex obtusifolius*, *Sonchus oleracea, Taraxacum officinale, Trifolium repens, Myosotis arvensis* and *Urtica dioica*. Glyphosate herbicide (Roundup® Pro Biactive®, contains 360 g/L glyphosate and is recommended for the control of annual and perennial grasses and broad-leaves weeds) was applied to the existing vegetation in the trial area and the retention pond at 4.0 L per hectare (Hitchmough et al., 2008) every two months from August 2012 until October 2013 to remove existing vegetation.

A shallow swale was excavated in the trial area in July 2012, which was then divided into 15 sections to enable vegetation trials in rain garden conditions. The size of each experimental unit was 2.0 $\times$ 3.4 m, with 0.35 m spacing between replicates (Fig. 5.7a). The 15 experimental units were excavated to have their depression bottom areas on the same level (Fig. 5.7b), so that hydrological connection was established amongst the 15 experimental units to ensure the water input could evenly contribute into each experimental unit. Each experimental unit has two 400 mm wide strips as part of its marginal area on both sides of its depression (Fig. 5.8). During the whole study, the hedgerow on the north of the experimental units was pruned to keep a 200 mm distance away from the edges of the 400 mm wide marginal strips (Fig. 5.8). There was a 0.5 m wide maintenance path besides the orchard field on the south of the 400 mm wide marginal strips at the south sides of the experimental units (Fig. 5.8). The depression of each experimental unit was properly excavated to have 1:3 (1 vertical to 3 horizontal) side slopes, and the depth of depression was 400 mm (Fig. 5.8).

In order to reduce the runoff velocity to prevent flow-induced erosion in a severe wet weather, a 300 mm wide check dam was established between every three experimental units. Fig. 5.7b shows the longitudinal section of the positions of the check dams and the locations of the 15 experimental units. The check dams were constructed with bricks and had three small drainage holes (30 mm diameter) (Fig. 5.9). There was a 75 mm vertical distance from the drainage holes in each check dam to the level of depression bottom that formed the retention depth of the experimental units (Fig. 5.8). The drainage holes could allow excess water to flow through at a relatively slow rate to allow hydrological connection amongst the experimental units.

The positions of the different saturation zones of the margin, slope and bottom in the experimental units were given in Fig. 5.7c. In this study, the defined margin of each experimental unit consisted of the marginal strips and angular surfaces with 75 mm vertical height to the ground level on both sides of the experimental unit (Fig. 5.8). The bottom of each experimental unit was defined to include the lowest depression base and angular surfaces with 75 mm vertical height on both sides of the lowest base (Fig. 5.8).

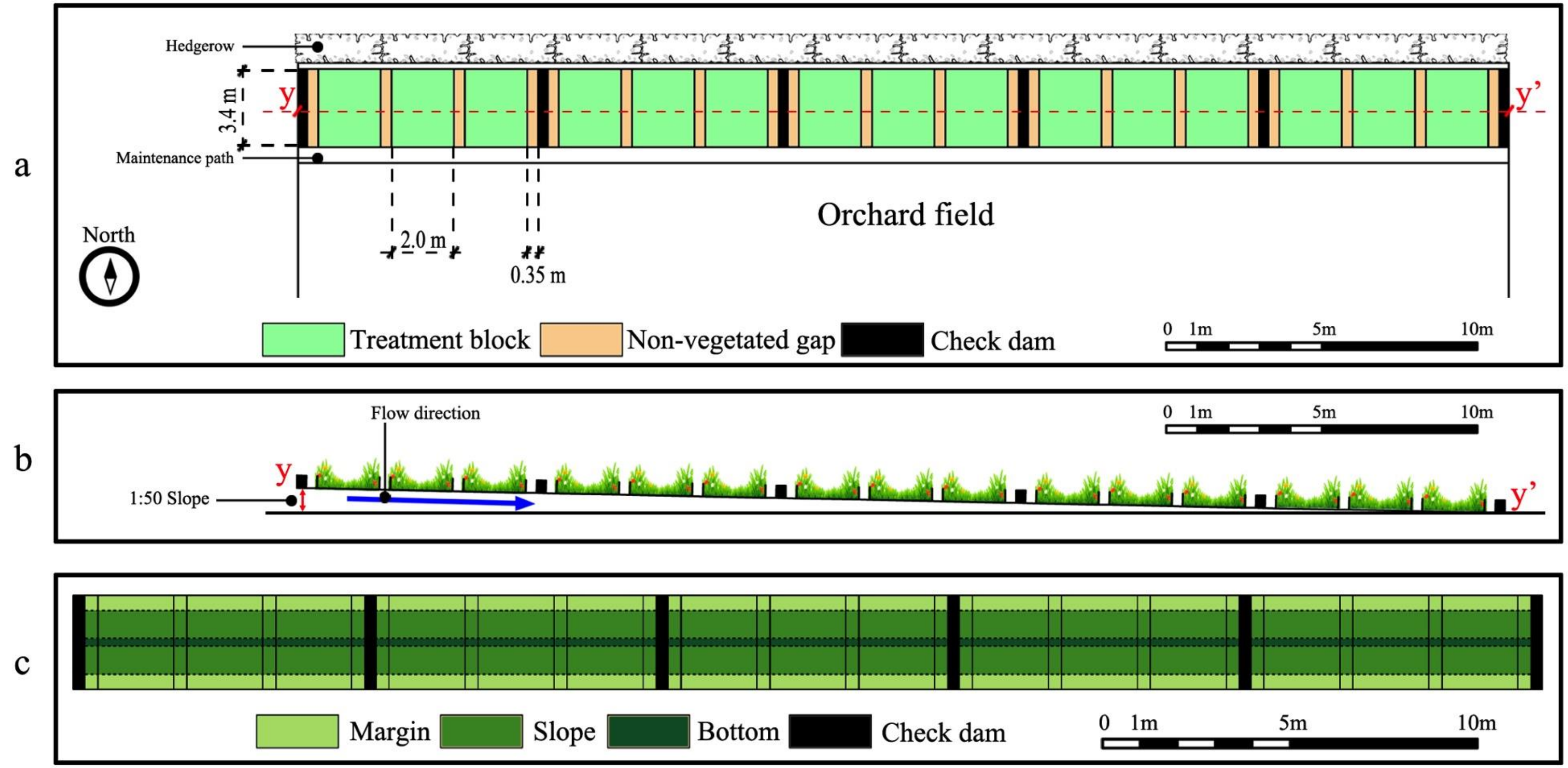


Fig. 5.7. Illustrated diagram of the 15 experimental units and different saturation zones in the experimental units

a: Schematic illustration of the layout of the 15 experimental units and their locations within the study site.

b: Longitudinal section showing the position of the check dams and each of the experimental units.

c: Schematic illustration indicating the position of the 'margin-slope-bottom' saturation zones in the experimental units.

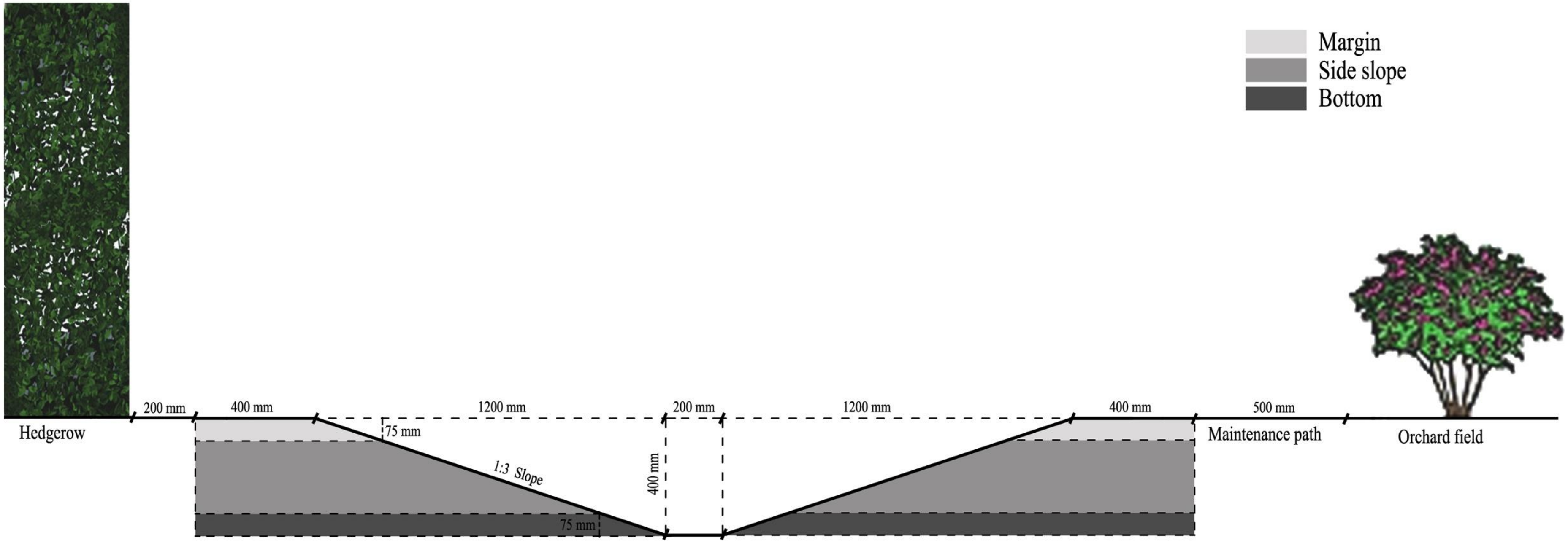


Fig. 5.8. Sectional illustration of each experimental unit

Fig. 5.9. Check dam with drainage holes at bottom. Photo was taken by the author in August 2014.

Soil samples taken in-situ were sent to the Department of Geography, University of Sheffield on 30 July 2013 for analysis to obtain the physical characteristics (e.g. percentage content of sand, silt and clay in the substrate, and porosity of the soil) and chemical characteristics (e.g. percentage content of organic matter and pH). The initial soil texture was classified as a loam (30.6% sand, 44.5% silt and 13% clay) with 10.7% Pebble (>4 mm), 1.35% Gravel (2-4 mm) and a porosity of 43.1%. The existing soil had an organic matter content of 6.2% and a pH of 6.3. The soil infiltration rate was measured on-site on 12 March 2014. The method of the measurement of onsite soil infiltration rate was the same as presented previously in Chapter 3. It was neither free-draining nor easily waterlogged, with an infiltration rate of 4.1 cm/hour. It met the requirements for water infiltration and planting establishment in typical rain garden applications, which was recommended from 0.25 cm/hour in clay loam soil to 21 cm/hour in sandy loam soil (Woelfle-Erskine & Uncapher, 2012).

5.2.2 Plant selection

Fifteen British native perennial species including fourteen flowering forbs and one grass were used in this study to produce the seed mix (Table 5.1). It should be noted that the overall approach in this thesis for plant selection for typical rain gardens is to create diverse mixes, without specific regard to nativeness. However, the goal of the present study is purely to look at the effects of the two simple weed control applications (i.e. mulch depth and the use of felt mat) and the potential ‘dry-wet’ moisture gradient in the experimental units on the emergence of the sown plantings. Therefore, in this study, only British native species were chosen for their availability in the British market, for their reasonable price to cope with the funding limitation, and for their potential to be used as indicator species.

In consideration of the aesthetic values of the sown forb-rich community, the fourteen native forbs were chosen to go into this specific mix either for their significant flowering from late spring to summer and the bright colours of their flowers to have a sharp contrasting mix of colour (e.g. using *Leucanthemum vulgare* for its clearly visible white flowers to contrast with the purplish pink flowers *Lychnis flos-cuculi* in late spring and early summer) (Fig. 5.10), or for their different shapes and textures in stems and leaves and various heights in plants to provide the complexity in community structure and the interest of wildness (e.g. using *Filipendula ulmaria* for its irregularly pinnate lance-shaped leaves, and using

Deschampsia cespitosa for this grass species' linear, ever-green leaves and its spikelet from early to late summer) (Fig. 5.11).

Fig. 5.10. The mid-June bright contrasting colour mix provided by the white flowers of *Leucanthemum vulgare* and the pinky flowers of *Lychnis flos-cuculi* from the experimental plots. Photo was taken by the author in June 2015.

Fig. 5.11. Spikelets of *Deschampsia cespitosa* added different shapes and texture into the sown community for the interest of wildness. Photo was taken by the author in June 2015.

Table 5.1 indicates the visual and aesthetic character of this mix, i.e. the flowering periods of each species to indicate when the mix will be flowering, and the approximate heights and spreads of each species to indicate the structure of the mix. In terms of the expected flowering display in this specific forb-rich mix, the first species to flower would be *Cardamine pratensis* (purple or lilac) in late April, while the first wave of flowers commenced in May with *Achillea millefolium* (white), *Cardamine pratensis* (purple or lilac), *Galium mollugo* (white), *Leucanthemum vulgare* (bright white) and *Rumex acetosa* (reddish-green). The second wave of flowers commenced in June with *Achillea millefolium* (white and pink), *Centaurea nigra* (bright purple), *Deschampsia cespitosa* (silver-tinted purple spikelets), *Galium mollugo* (white), *Leucanthemum vulgare* (bright white), *Lotus pedunculatus* (golden-yellow), *Lychnis flos-cuculi* (pale to purplish pink), *Rumex acetosa* (reddish-green), *Thalictrum aquilegifolium* (purple-pink) and *Valeriana officinalis* (pink or white). The mix would reach its peak flowering display in July when all the June dominated species would keep their flowers, while the flowers of *Filipendula ulmaria* (creamy white), *Lythrum salicaria* (purple-pink), *Prunella vulgaris* (violet-blue) and *Succisa pratensis* (white or pink) would add into this mix. Flowering species would decrease in August compared with that in June, which would be dominated by *Achillea millefolium* (white and pink), *Centaurea nigra* (bright purple), *Deschampsia cespitosa* (silver-tinted purple spikelets), *Filipendula ulmaria* (creamy white), *Galium mollugo* (white), *Lythrum salicaria* (purple-pink), *Prunella vulgaris* (violet-blue), *Rumex acetosa* (reddish-green), *Succisa pratensis* (white or pink) and *Valeriana officinalis* (pink or white). Flowering species would further decrease in September, with only four species *Centaurea nigra* (bright purple), *Galium mollugo* (white), *Prunella vulgaris* (violet-blue) and *Succisa pratensis* (white or pink) continuously providing flowers. In October, only *Succisa pratensis* (white or pink) flower. The sown forb-rich mix is expected to have approximately 3 months (June, July and August) of dramatic flowering display (Table 5.1). The various heights of plants may provide a ‘see-through’ effect through the architecture of the sown community, in which the species with larger flowers and significant colour (e.g. *Leucanthemum vulgare* and *Lychnis flos-cuculi*) will provide a sharply contrasting colour mix, surrounded by the multi-hued smaller flowers of other species (Fig. 5.12).

Table 5.1. Plant type, approximate height and flowering phenology of the sown species. (Adapted from Brickell, 2008; Dunnett & Clayden, 2007; Hansen & Stahl, 1993; Hubbard, 1984; Steiner & Domm, 2012; Thomas, 1976.)

Species	**Plant type**	**Approx. Height (m)**	**Approx. Spread (m)**	Jan	Feb	March	April	May	June	July	Aug	Sept	Oct	Nov	Dec
Achillea millefolium	Forb	0.1-0.5	0.1-0.5					■	■	■	■				
Cardamine pratensis	Forb	0.1-0.5	0.1-0.5				■	■							
Centaurea nigra	Forb	0.5-1.0	0.1-0.5						■	■	■	■			
Deschampsia cespitosa	Grass	0.5-1.2	0.5-1.0						■	■	■				
Filipendula ulmaria	Forb	0.5-1.0	0.5-1.0							■	■				
Galium mollugo	Forb	0.1-1.0	0.1-0.5					■	■	■	■	■			
Leucanthemum vulgare	Forb	0.5-1.0	0.1-0.5					■	■	■					
Lotus pedunculatus	Forb	0.1-0.5	0.1-0.5						■	■					
Lychnis flos-cuculi	Forb	0.5-1.0	0.1-0.5						■	■					
Lythrum salicaria	Forb	1.0-1.5	0.1-0.5							■	■				
Prunella vulgaris	Forb	0.1-0.5	0.1-0.5							■	■	■			
Rumex acetosa	Forb	0.5-1.0	0.1-0.5					■	■	■					
Succisa pratensis	Forb	0.5-1.0	0.1-0.5							■	■	■	■		
Thalictrum aquilegifolium	Forb	0.5-1.0	0.1-0.5						■	■					
Valeriana officinalis	Forb	1.0-1.5	0.5-1.0						■	■	■				

■ Plant in flower

Fig. 5.12. Flowering display dominated by *Leucanthemum vulgare* in early June in the sown forb-rich community. Photo was taken by the author in June 2015.

Further typical characteristics (e.g. stem, leaflets, flowers and habits, etc.) of each species used in this study and the typical habitats of these species are given below, following the guide of a series of botanic documents and rain garden specific manuals (Brickell, 2008; Dunnett & Clayden, 2007; Hansen & Stahl, 1993; Hubbard, 1984; Steiner & Domm, 2012; Thomas, 1976). Images of each species are also provided. However, a few species had no plant showing their full blooms during the limited time of this study, the images of these species are thus adapted from other sources including "The Royal Horticultural Society A-Z encyclopaedia of garden plants" (Brickell, 2008) and the official website of the Royal Horticultural Society (https://www.rhs.org.uk).

Achillea millefolium Rhizomatous, low growing and mat-forming perennial. Pinnatisect leaves are linear to lance-shaped. It produces flower heads borne in flat corymbs. It is naturally found in meadows and pastures in moist but well-drained soil in full sun, but can tolerate a wide range of soils and moisture conditions.

Fig. 5.13. Image of *Achillea millefolium***.** Photo was taken by the author in August 2014.

Cardamine pratensis Variable perennial with erect stems and rosettes pinnate, grey-green foliage. It bears panicles of purple, lilac, or white flowers reminiscent of stocks. It grows in humus-rich, moist soil in full-sun or partially shady meadows and by streams.

Fig. 5.14. Image of *Cardamine pratensis***.** Source, Brickell, 2008.

Centaurea nigra Perennial with deeply lobed and hairy leaves. It produces flower heads with conspicuous involucre, overlapping bracts with black tips, and bearing bright purple flowers. It is naturally found in well-drained grassland, and grows in full sun and tolerates some drought.

Fig. 5.15. Image of *Centaurea nigra.* Photo was taken by the author in September 2014.

Deschampsia cespitosa Dense, tussock-forming, evergreen grass with rigid, linear, rough and mid-green leaves. It produces airy, arching panicles of glistening, silver-tinted purple spikelets. It naturally grows in meadows and floodplain of rivers and streams in sun or partial shade, and thrives in a wide range of moisture from dry to damp soil.

Fig. 5.16. Image of *Deschampsia cespitosa.* Source, the official website of the Royal Horticultural Society.

Filipendula ulmaria Clump-forming perennial with leafy stems bearing irregularly pinnate strongly veined lance-shaped leaves. Its branching stems bear dense corymbs of creamy white flowers. It naturally grows in wet ground in swamps, marshes, wet woods and meadows with moist soil, and prefers full sun or partial shade.

Fig. 5.17. Image of *Filipendula ulmaria.* Source, the official website of the Royal Horticultural Society.

Galium mollugo Rhizomatous perennial with whorls of linear leaves and rough, weak stems. It produces white tubular flowers with four spreading and narrow lobes. It naturally grows in hedgebanks, open woodland, scrub and grassy slopes, and may thrive in a wide range of soil conditions, preferably moist, humus-rich soil in sun or partial shade.

Fig. 5.18. Image of *Galium mollugo.* Photo was taken by the author in June 2015.

Leucanthemum vulgare Variable, rhizomatous perennial with obovate-spoon-shaped, toothed and dark-green basal leaves. It produces solitary flower heads with bright yellow disc-florets and white ray-florets. It grows in moderately fertile, moist but well-drained meadows or grassy fields in full sun or partial shade.

Fig. 5.19. Image of *Leucanthemum vulgare.* Photo was taken by the author in June 2015.

Lotus pedunculatus Erect perennial with hollow stems and sepals turn back at their tips. It produces golden-yellow flowers borne in an umbel at the tip of the stem. It naturally grows in damp meadows and marshes, and prefers moderately fertile soil in full sun.

Fig. 5.20. Image of *Lotus pedunculatus.* Photo was taken by the author in July 2014.

Lychnis flos-cuculi Slender, sparsely hairy perennial with inversely lance-shaped, bluish green, oblong-lance-shaped and stem-clasping leaves. It produces loose, few-flowered, branched, terminal cymes of star-shaped, pale to purplish pink flowers. It often grows in damp meadows, marshes and wet woodlands, and prefers moderately fertile soil in full sun or partial shade.

Fig. 5.21. Image of *Lychnis flos-cuculi.* Photo was taken by the author in June 2015.

Lythrum salicaria Clump-forming perennial with erect branched stems bearing lance-shaped, downy leaves. It produces star-shaped, bright purple-red to purple-pink flowers in spike-like racemes. It prefers full sun and naturally grows in swamps at the margins of lakes and slow-flowing rivers and marshes.

Fig. 5.22. Image of *Lythrum salicaria.* Source, the official website of the Royal Horticultural Society.

Prunella vulgaris Low, spreading perennial bearing ovate to diamond-shaped, deep-green leaves. Leafy stems produce whorls of violet-blue, sometimes white to pink flowers. It naturally grows in waste ground, grassland, woodland edges with moderately fertile and moist soil in sun or partial shade.

Fig. 5.23. Image of *Prunella vulgaris.* Photo was taken by the author in July 2014.

Rumex acetosa Slender perennial bearing edible, arrow-shaped leaves. It produces whorled spikes of reddish-green flowers in early summer, which then turn purplish. It grows in moist to damp soil in full sun or partial shade, and is naturally found in meadows by streams and open places in woodland.

Fig. 5.24. Image of *Rumex acetosa*. Photo was taken by the author in July 2014.

Succisa pratensis Rosette-forming, rhizomatous perennial with thin, branched, slightly hairy stems and elliptic basal leaves. It bears solitary, pincushion-like, violet, sometimes white or pink flower heads. It grows in poor to moderately fertile, peaty soil, and naturally occurs in meadows, pastures and marshes. It thrives in moist to damp in full sun or partial shade.

Fig. 5.25. Image of *Succisa pratensis.* Source, the official website of the Royal Horticultural Society.

Thalictrum aquilegifolium Erect, clump-forming stems bearing 2- or 3-pinnate, hairless leaves, which are composed of obovate, wavy-margined leaflets. It produces clustered, fluffy flowers with greenish white sepals, bright purple-pink or white stamens, and flat-topped, terminal panicles on glaucous stems. It grows in moist, humus-rich soil in partial shade.

Fig. 5.26. Image of *Thalictrum aquilegifolium.* Photo was taken by the author in July 2015.

Valeriana officinalis Upright, clump-forming perennial with short rhizomes producing branching stems and aromatic, bright green, pinnate, basal leaves with lance-shaped, toothed leaflets. It bears branched, rounded, corymb-like cymes of pink or white flowers in summer. It grows in grassland or scrubs on dry or moist soils in full sun or partial shade.

Fig. 5.27. Image of *Valeriana officinalis.* Source, the official website of the Royal Horticultural Society.

The preferred hydrological regimes (i.e. the duration, frequency, timing and predictability of the flooded and dry phases, Bunn et al., 1997) of the sown species are listed in Table 5.2, as well as the differentiation in their assumed distribution in the ‘margin-slope-bottom’ rain garden profile. In general, three levels of hydrological regime are indicated, which range from: (1) continuous inundation (i.e. ‘wetland’ species), (2) periodic or seasonal inundation (i.e. species from wet meadows or other habitats that are not permanently wet), and (3) infrequent inundation (i.e. species from fertile habitats in temperate maritime climates). The assumed hydrological regimes of the sown species in this study are based on the instructions of a series of botanic and rain garden specific documents (Brickell, 2008; Dunnett & Clayden, 2007; Hansen & Stahl, 1993; Hubbard, 1984; Steiner & Domm, 2012; Thomas, 1976). The sown species in this study are identified as appropriate plant options for use in typical rain gardens, as most of these species could thrive in moist to damp soils.

Table 5.2. Assumed hydrological regimes of the sown species and their assumed distribution throughout the 'margin-slope-bottom' saturation zones in rain gardens. (Adapted from Brickwell, 2008; Dunnett & Clayden, 2007; Hansen & Stahl, 1993; Hubbard, 1984; Steiner & Domm, 2012; Thomas, 1976.)

Species	Assumed hydrological regime	Margin	Slope	Bottom
Achillea millefolium	Periodic or seasonal inundation	*	*	
Cardamine pratensis	Periodic or seasonal inundation	*	*	
Centaurea nigra	Infrequent inundation	*	*	
Deschampsia cespitosa	Periodic or seasonal inundation	*	*	*
Filipendula ulmaria	Periodic or seasonal inundation		*	*
Galium mollugo	Periodic or seasonal inundation	*	*	
Leucanthemum vulgare	Infrequent inundation	*	*	
Lotus pedunculatus	Continuous inundation		*	*
Lychnis flos-cuculi	Periodic or seasonal inundation	*	*	
Lythrum salicaria	Continuous inundation	*	*	*
Prunella vulgaris	Infrequent inundation	*	*	
Rumex acetosa	Periodic or seasonal inundation	*	*	
Succisa pratensis	Periodic or seasonal inundation	*	*	
Thalictrum aquilegifolium	Infrequent inundation	*	*	
Valeriana officinalis	Infrequent inundation	*	*	

* Assumed distribution

5.2.3 Production of the seed mixture and lab germination test of the sown species

In this study, fresh seeds produced in summer and autumn 2013 were used, which were obtained from Emorsgate Seeds Ltd. (Norfolk, UK). Seeds of all the selected species were obtained in early October 2013, which were then mixed on the basis of seed weight (Table 5.3). It is worth noting that the species composition was expressed as percentage weight in table 3 rather than giving the actual or assumed species composition for the germination in the field. A portion of seeds of different species were used for a germination test, which was carried out under laboratory conditions on 12 October 2013 in Green Estate Ltd., Sheffield, UK. In this test, 50 seeds per species were placed in a Petri dish with moist blotting paper to observe their germination in the lab at 20 ℃. Each species had two dishes (i.e. 100 seeds), which were checked with a one-day interval. Germination experiment in each species was allowed at least 30 days after sowing seeds in Petri dishes, and was terminated when there was no germination for at least one week. At the termination of test, the germinated seedlings of each species were counted and then the germination rate was calculated. The results of the germination tests are given in Table 5.3.

Table 5.3. Selected species and their germination results under lab conditions. Species composition was expressed as percentage weight.

Species	% Weight	Lab germination (%)
Achillea millefolium	5	90
Cardamine pratensis	5	40
Centaurea nigra	5	70
Deschampsia cespitosa	10	78
Filipendula ulmaria	10	4
Galium mollugo	5	60
Leucanthemum vulgare	5	90
Lotus pedunculatus	5	45
Lychnis flos-cuculi	10	46
Lythrum salicaria	5	40
Prunella vulgaris	5	62
Rumex acetosa	5	90
Succisa pratensis	10	62
Thalictrum aquilegifolium	10	82
Valeriana officinalis	5	65

5.2.4 Experimental design

5.2.4.1 Treatment set up

In this study, cultivation of site was carried out two weeks before the treatments were applied, so that the treatments could be implemented into a fine tilth. Five different treatments were implemented on 17 December 2013. Each treatment was replicated three times. The five treatments were varied by the depth of green waste compost mulching and by the presence of felt mats (with or without). The purpose of using the mulch was to give the sown seeds a head start so that they might have advantages in competition, while the felt mats were assumed to mitigate the growth of vegetative fragments of weeds in soils. The green waste compost was obtained from Green Estate Ltd., Sheffield, UK. The compost was made according to the British Standard for compost making (WRAP, 2002), which used stockpiling food and yard waste such as plant leaves. The analysis results of the green waste compost were reported on 3 October 2013 by Alliance Technical Laboratories Ltd. (Suffolk, UK), in which the used compost was claimed to be weed free and the main properties of the compost mulch had a bulk density of 0.566 $g \cdot cm^{-1}$ with 71.2% organic matter, 41.4% organic carbon and a pH of 8.4. To further verify whether the green waste compost was weed free, a sample test was carried out on 11 October 2013. 20 L of the compost was placed into ten 2 L pots. The ten pots were placed in a greenhouse at a nursery area located in Green Estate Ltd., Sheffield, UK. The pots were irrigated once in two days for 30 days, and no seedling emergence was found. The felt mats were obtained from Lindum Turf (York, UK). As noted previously, the felt mats were made from recycled cotton and wools, which aimed to mitigate the emergence of unexpected weedy species in existing soils.

Treatments consisted of:

1. Control group: a non-treated control group in which seeds were directly sown and raked into the existing soil bed. The sowing rate of the seed mix was 2 g per m^2. The seeds for each experimental unit were evenly mixed with 2 L sharp sands (grain sizes ranging from 0.06 to 4.0 mm, obtained from B&Q, Eastleigh, UK) and then evenly sown into each block.

2. Group A: a group with felt mats laid above the existing surface (Fig. 5.28). Compost was mulched on top of these mats to a depth of 20 mm. Seeds were then sown with the same method as noted previously, and were raked into the mulched compost.

3. Group B: a group with felt mats laid above existing surface. Compost was mulched on top of these mats to a depth of 100 mm. Seeds were then sown with the same method as noted previously, and were raked into the mulched compost.

4. Group C: a group with compost mulched over existing surface to depth of 20 mm without the use of felt mats. Seeds were then sown with the same method as noted previously, and were raked into the mulched compost.

5. Group D: a group with compost mulched over existing surface to a depth of 100 mm without the use of felt mats. Seeds were then sown with the same method as noted previously, and were raked into the mulched compost.

Fig. 5.28. Spreading green waste compost on the top of the felt mat laid on the existing soil. Photo was taken by the author in December 2013.

It is worth noting that there was no group adopting felt mat with 0 mm compost mulching, because mulching must be applied on top of felt mat to form sowing bed for seed mixes. The layout of the five different treatments and their replicates is shown in Fig. 5.29. It is noticeable that the sowing was carried out outside of the normal recommended sowing times for meadows (i.e. in early autumn for sowing herbaceous perennials, Brickell, 2002). Sowing was delayed in this study because the felt mats were delivered to the site on 15 December 2013. However, the average temperature in December 2013 was 6.3 ℃ without snow events (Met Office, 2015), and was thus relatively acceptable for sowing. A benefit of sowing in December is that there will be no or rather little wind-born seeds to contaminate the plots. Also, some seeds might benefit from winter chilling before germination in the next year. In order to further prevent flow-induced erosion, hessian erosion control mats were pinned over the top in every experimental unit to hold soil in place after seeds were sown and raked into the substrates (Fig. 5.30).

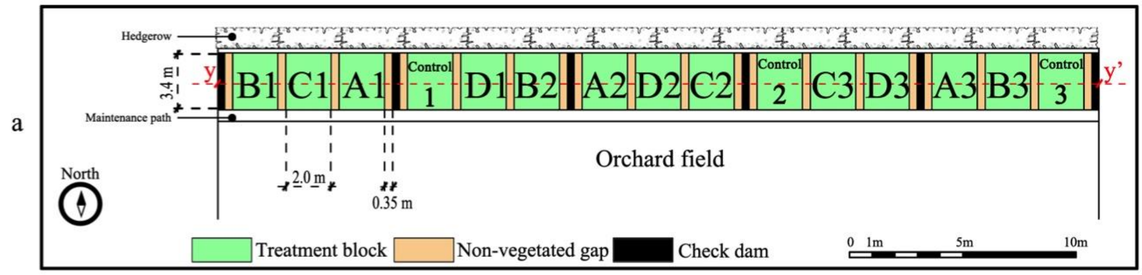


Fig. 5.29. Layout of the five different treatments and their replicates.

Fig. 5.30. Hessian erosion control mat pinned over the top in every experimental unit to hold soil in place. Photo was taken by the author in December 2013.

The forb-rich mixes were allowed to grow naturally on-site for about nine months before counting the emergence of the sown species. During this period, the plants relied on natural precipitation without any artificial irrigation. No maintenance (e.g. weeding, mowing or cutting back, etc.) was given during the whole duration of this study, so that the trials enabled the emergence of the sown community responding to the actual situation in the experimental units to be monitored. Due to the funding limitation for the present study, there was no device that could be used to monitor the weather conditions during the nine months. Therefore, the monthly average temperature and the monthly rainfall during the nine months were from Met Office (2015) (Fig. 5.31). The Sheffield's official Met Office recording station is located at Weston Park, Sheffield, UK (53°22'53.0"N, 1°29'29.0"W), less than 4 km east of the study rain gardens. In this study, the average temperature in every month within the germination period and the monthly rainfall was compared with the average data provided from 1955 to 2015 (Met Office, 2015). The winter was relatively warm with slightly higher average temperature in December 2013, January and February 2014 (6.3 ℃, 5.0 ℃ and 5.6 ℃, respectively) compared with the average December, January and February temperature from 1955 to 2015 (4.5 ℃, 3.8 ℃ and 3.9 ℃, respectively) (Met Office, 2015). December 2013 was a relatively dry month with 65.5 mm monthly rainfall, which was below the average (86.5 mm) from 1955 to 2015 (Met Office, 2015). However, monthly rainfall in January and February 2014 (137.5 mm and 106.4 mm) was significantly greater than the average from 1955 to 2015 (81.3 mm and 63.8 mm) (Met Office, 2015), which indicates a very wet winter in early 2014. There was a warm spring in 2014, with the average temperature in March, April and May 2014 (7.4 ℃, 10.1 ℃ and 12.2 ℃, respectively) above the average from 1955 to 2015 (5.8 ℃, 8.2 ℃ and 11.4 ℃, respectively) (Met Office, 2015). The early spring in 2014 was relatively dry, with rainfall in March 2014 (49.0 mm) which was much less than average (62.9 mm), while April monthly rainfall in 2014 (63.0 mm) was slightly higher than average (60.0 mm) (Met Office, 2015). However, May 2014 was very wet, with May monthly rainfall (139.5 mm) more than twice the average from 1955 to 2015 (60.9 mm) (Met Office, 2015). Average temperature in June and July 2014 (15.3 ℃ and 17.9 ℃) was above average from 1955 to 2015 (14.3 ℃ and 16.2 ℃), while August 2014 (14.9 ℃) was slightly cooler than average (15.9 ℃) (Met Office, 2015). Early summer in 2014 was a very dry period with much less monthly rainfall in June and July (49.6 mm and 34.3 mm) than the average from 1955 to 2015 (67.1 mm and 60.6 mm) (Met Office, 2015). However, August 2014 (92.6 mm) was wetter than average (67.8 mm) (Met Office, 2015).

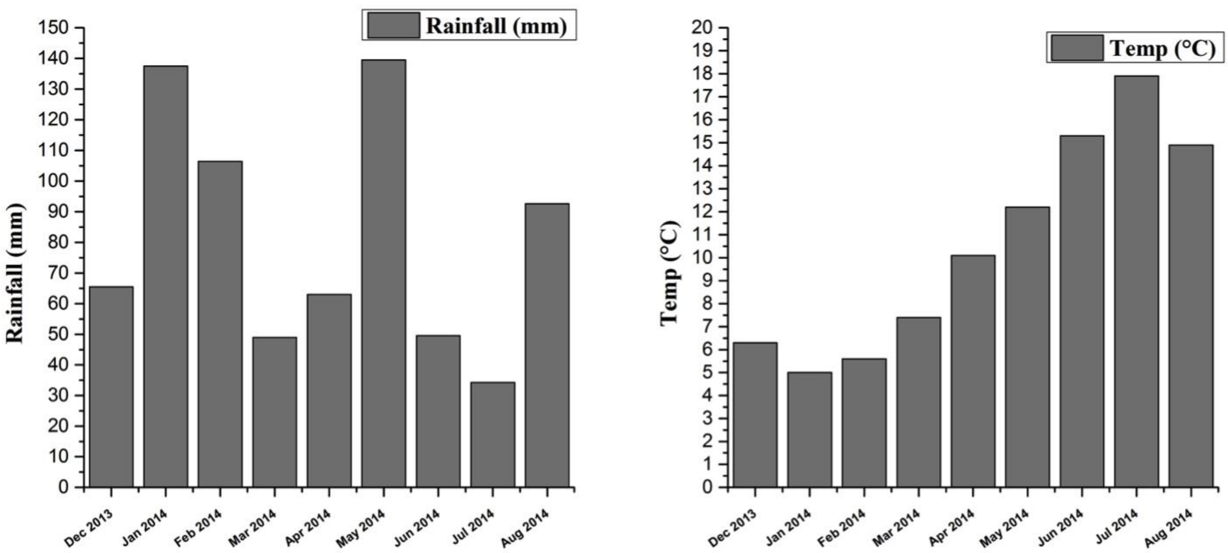


Fig. 5.31. Monthly average temperature and monthly rainfall recorded at Sheffield's recording station, Weston Park, Sheffield, UK.

5.2.4.2 Observation of plant emergence in the experimental units

On 19 September 2014, the number of sown and unsown ruderal species and the number of seedlings of sown species and unsown ruderal species, as well as the % coverage of vegetation includes all sown species and unsown weeds from the bottom, side slope and margin on the south bank of each experimental unit were recorded. Three replicate quadrats were assessed using a randomly distributed 0.5 × 1.0 m quadrat (leaving a buffer of at least 50 mm from the edge) in the margin, slope and bottom from each experimental unit, respectively, and calculated the mean value. In this study, % cover of the vegetation was estimated visually among the treatments. % cover from the margin, slope and bottom of each experimental unit were also estimated visually in the randomly distributed 0.5 $\times$ 1.0 m quadrat for three measurements per treatment or per saturation zone and calculated the mean value.

The reason to obtain data from only the rain garden bottoms and the slopes and margins in the south side of the experimental units was because these saturation zones were in full sun without the influence of the hedgerow shade (Fig. 5.32). The hedgerow shade made the north margin and north slopes of each experimental unit an area with heavy shading (Fig. 5.32). As noted previously, all the selected sown species would prefer full sun or partial shade. Germination of herbaceous plants is reliant upon at least moderate exposure to light (Gabriell, 1997), thus the emergence of the sown species in the north banks of the experimental units would be greatly altered by the hedge shade. Moreover, the heavy shading in the north banks of the experimental units would alter the temperature and thus potentially influence the evapotranspiration, so that soil moisture in the north banks of the experimental units might be altered.

Fig. 5.32. The north margin and slope was shady in the experimental units, while the bottom and the south side of the experimental units were in full sun. Photo was taken by the author in August 2014.

5.2.4.3 On-site observation of soil moisture distribution in the 'margin-slope-bottom' rain garden profile

From 17 December 2013 to 16 September 2014, the trials were checked weekly. During this period, on-site observations found that the margins of all experiment units appeared to be the driest zones most of the time, while slopes were found to stay moderately moist which neither waterlogged and dried out (Fig. 5.33). Bottoms were found constantly damp and easily waterlogged in wet weather, which required multiple days to completely dewater due to no application of soil amendment and underlying drainage pipes (Fig. 5.34). However, the experimental rain gardens never got truly anaerobic or encountered severe drought during the experimental season.

Fig. 5.33. Moderate moist slopes and the drier margins in the experimental units. Photo was taken by the author in April 2014.

Fig. 5.34. Common inundation in the bottoms of the experimental units during the wet weathers. Photo was taken by the author in June 2014.

To understand the effects of the dynamic soil moisture distribution in the 'margin-slope-bottom' rain garden profile on the emergence and % coverage of the sown mix, it was important to observe the potential differences in moisture conditions from the different saturation zones including the sunny bottoms and the sunny slopes and margins on the south banks of the experimental units. However, due to the lack of necessary equipments, the on-site soil moisture data was measured outside of the experimental period. Although the moisture conditions of the experimental units were instrumented after the vegetation data had been collected, the moisture data was still considered to be valuable for knowing the exact moisture distribution trend throughout the ‘margin-slope-bottom’ rain garden profile.

From 16 September 2014 to 18 December 2014 (94 days in total), the experimental units were instrumented to continually monitor soil moisture throughout the rain garden profile, as well as the on-site temperature and rainfall. Continuous readings of soil volumetric water content (VWC) throughout the ‘margin-slope-bottom’ saturation zones were obtained hourly by one soil moisture sensor (CS616, Campbell Scientific, Loughborough, UK) per saturation zone on the sunny south side of a randomly picked experimental unit (D2) and the non-vegetated gap next to the selected experimental unit (Fig. 5.35). The sensors were installed obliquely (30 ° from the soil surface) at the sampling depth of 100-150 mm (i.e. topsoil). The soil moisture sensors had the accuracy in the field at ± 0.02 $m^3 \cdot m^{-3}$ (Campbell Scientific, 2004). There were six soil moisture sensors installed on-site, C1, C2 and C3 were installed along the vegetated block from rain garden bottom to margin, respectively, while the sensors of C4, C5 and C6 were placed along the non-vegetated gap from bottom to margin, respectively

(Fig. 5.35). Precipitation was recorded hourly by a tipping bucket raingauge (ARG100, Campbell Scientific, Loughborough, UK), which had a resolution of 0.2 mm. The tipping bucket raingauge was placed on a 2 m^2 flat ground in the study site where existing vegetation was cleaned. The soil moisture and precipitation data was automatically collected by a datalogger installed on-site (CR1000, Campbell Scientific, Loughborough, UK).

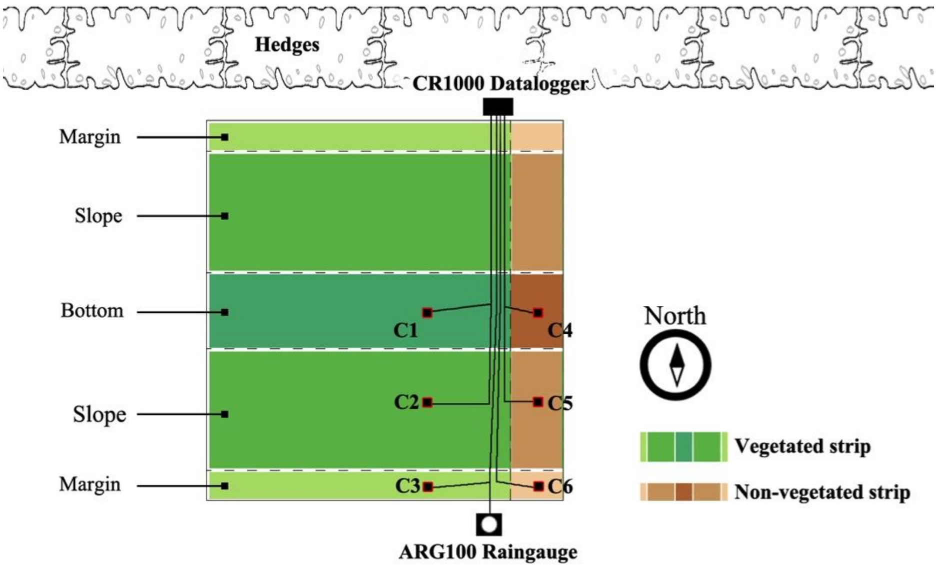


Fig. 5.35. Installation layout of soil moisture monitor instruments and tipping bucket raingauge

During this period, the mean air temperature was 9.2 ℃ with the minimum daily temperature of 1.2 ℃ on 14 December and maximum daily temperature of 16.8 ℃ on 19 September (Fig. 5.36). During this period, the on-site mean temperature in September (13.6 ℃) was the same as the average from 1955 to 2015 (Met Office, 2015). On-site mean temperature in October and November (11.1 ℃ and 8.0 ℃, respectively) were above average (10.3 ℃ and 6.6 ℃, respectively), while the on-site mean temperature (4.4 ℃) in December was the same as average (4.5 ℃) (Met Office, 2015).

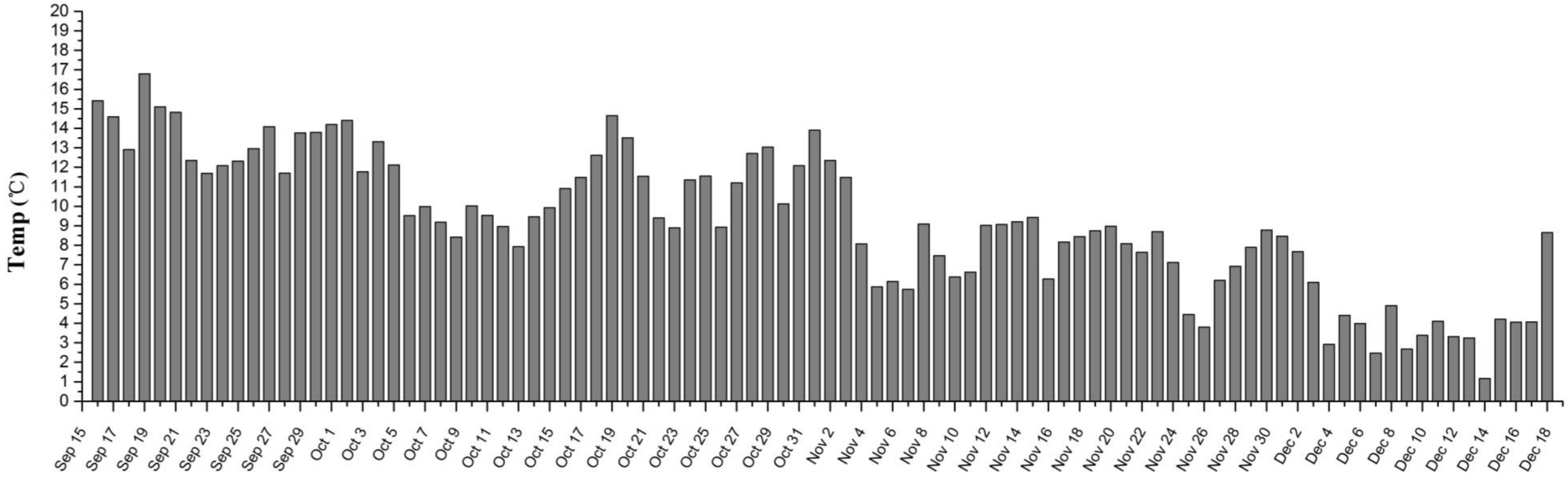


Fig. 5.36. On-site daily temperature from 16 September to 18 December 2014

During this period (16 September 2014 to 18 December 2014), rainfall and stormwater runoff from the orchard and outside site was the only available water input for the experimental units (Fig. 5.37). From 16 September 2014 to 18 December 2014, the total on-site rainfall over this duration was 240.0 mm with the recorded precipitation in September, October, November and December 7.6 mm, 86.6 mm, 102.2 mm and 43.6 mm, respectively. September 2014 was much drier than average from 1955 to 2015 (64.3 mm) (Met Office, 2015), despite the fact that the on-site rainfall was only obtained from 6 September. The on-site rainfall in October and November was higher than average (74.7 mm and 78.9 mm, respectively) (Met Office, 2015). The on-site rainfall in December was only recorded until 18 December, which was lower than average (86.5 mm) (Met Office).

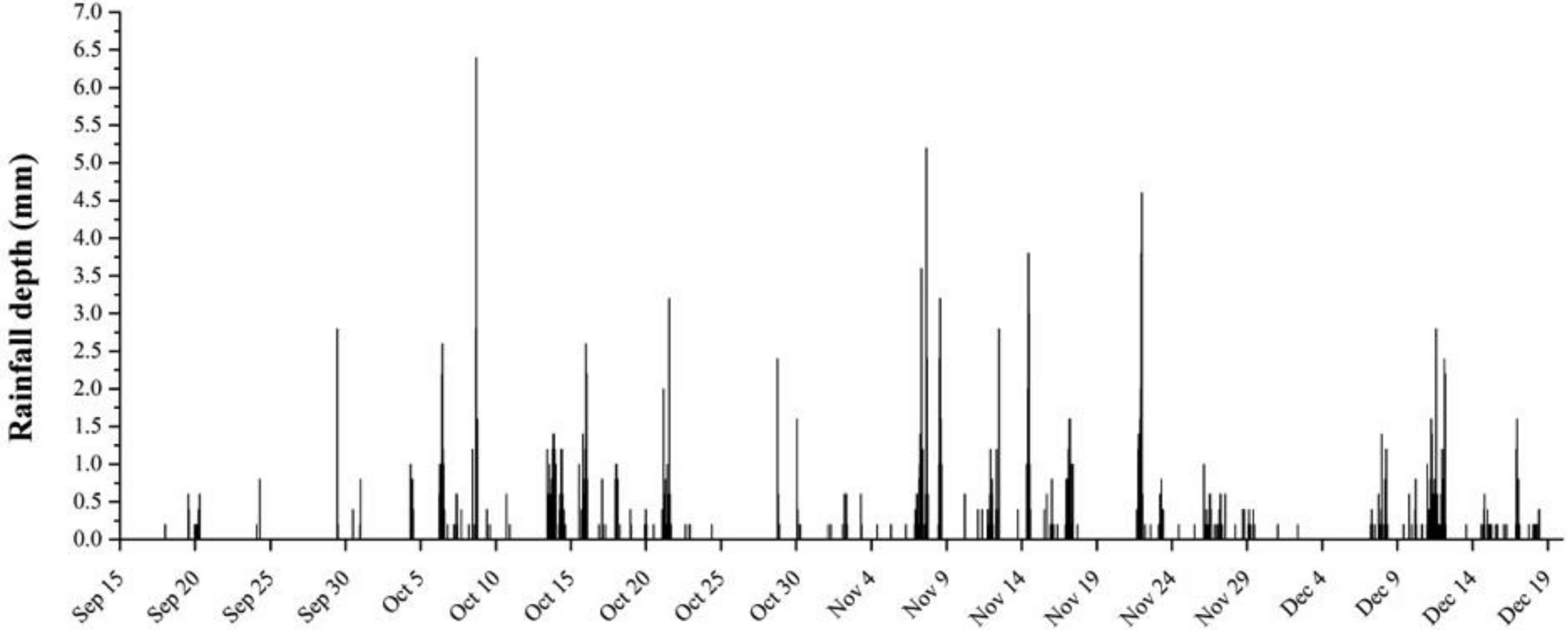


Fig. 5.37. Natural precipitation on site from 16 September to 18 December 2014

From 16 September 2014 to 18 December 2014, the only water input for the 15 experimental units was natural precipitation. The time-series plots (Fig. 5.38) show the dynamic changes of the soil moisture in the vegetated and non-vegetated rain garden profile. Soil volumetric water content (VWC) varied depending upon the size of preceding rainfall events. Generally, a consistently higher level of soil moisture was found in every saturation zone in the non-vegetative gap than in the vegetated plot. The soil moisture distribution throughout 'margin-slope-bottom' saturation zones in the non-vegetated gap was more complicated than in the vegetated plot. In the non-vegetative gap, soil moisture was greatest in depression bottom until it was exceeded by the VWC in margin from 6.00am, 11 December. The slope had higher VWC than in margin until 21.00pm, 21 October, after that soil moisture in margin was consistently higher than in the slope. Conversely, the vegetative plot consistently had the greatest VWC level in depression bottom, whereas the lowest level of soil moisture was consistently found in the margin.

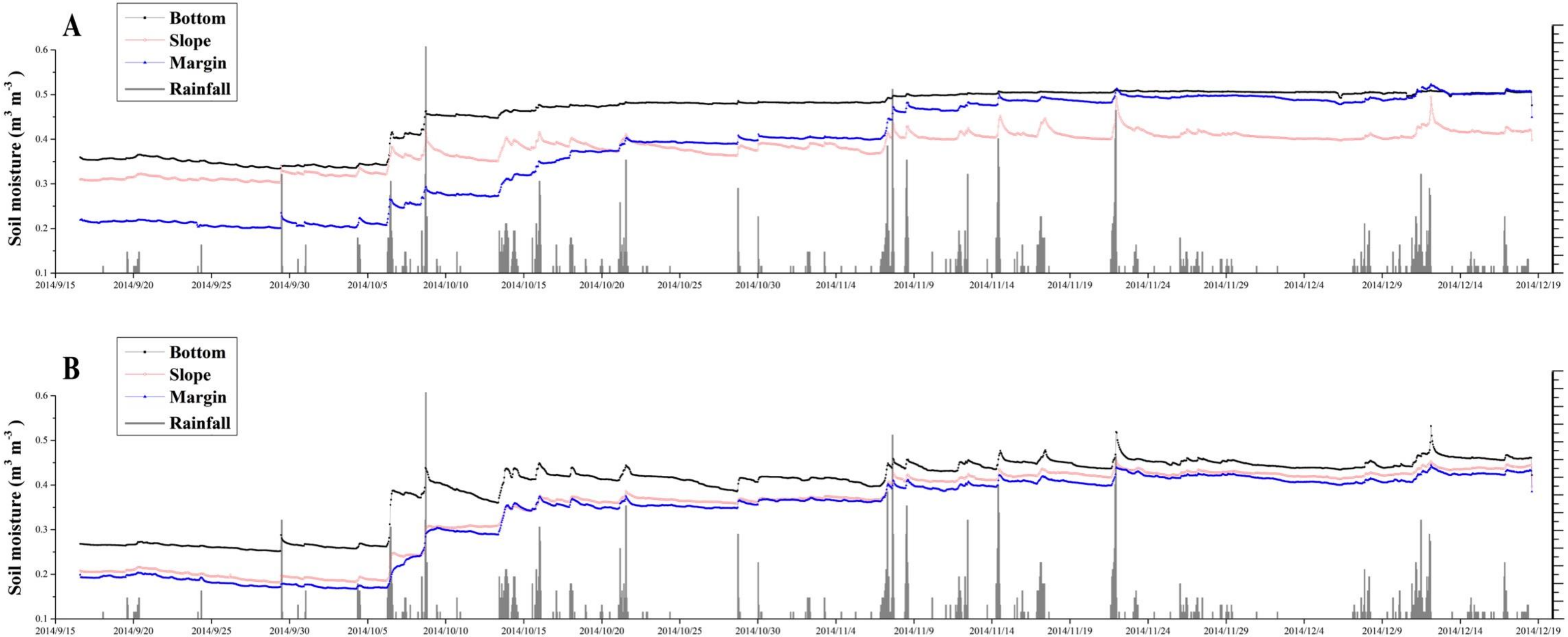


Fig. 5.38. Dynamic changes of the soil moisture throughout the rain garden profile (A: Non-vegetated gap; B: Vegetated experimental unit)

To further identify the soil moisture distribution throughout the 'margin-slope-bottom' rain garden profile, soil moisture data during the recording period was analysed using Two-way ANOVA in SPSS 20.0. The assumptions of normality in Two-way ANONVA were applied to the distribution of estimated values, not to the raw data. When dealing with a small data set, the estimated values' distribution is dependent on the distribution of the raw data. However, when a large amount of data was collected, the central limit theorem states that the distribution of estimates is approximately normal. Therefore, Two-way ANOVA was introduced to determine if the soil moisture in margin, slope and bottom was significantly affected by vegetation at two levels (with and without) and different saturation zones, considering the fact that a large amount of soil moisture data was measured on-site.

A two-way ANOVA showed that the soil moisture distribution during the period of 16 September to 30 September 2014 was significantly affected by different saturation zones (i.e. margin, slope and bottom) ($P<0.001$) and vegetative treatment (with or without) ($P<0.001$) (Fig. 5.39). The interaction between saturation zones and the presence of vegetative treatment was significant ($P<0.001$). Generally, the bottom was the wettest zone with the greatest mean value of VWC, followed by slope and margin (Fig. 5.39). Non-vegetated gap had greater VWC mean value in every saturation zone than that of the vegetated experimental unit.

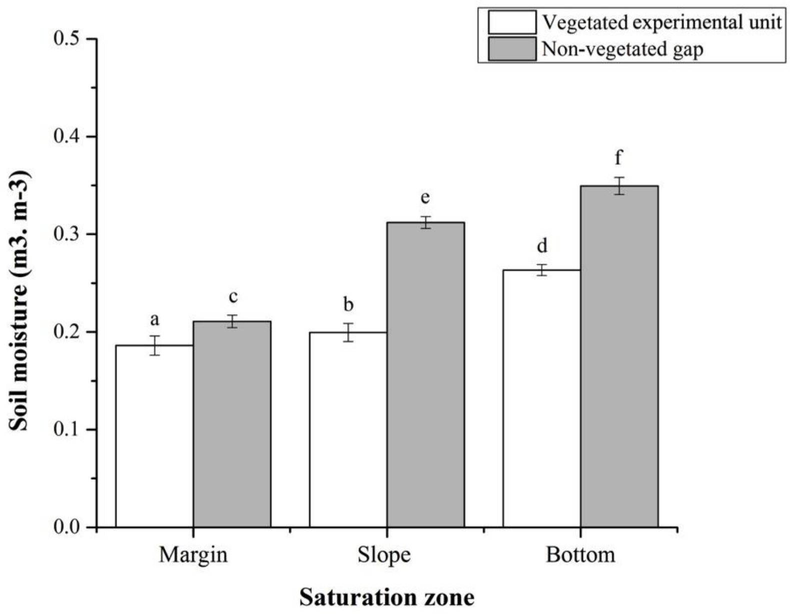


Fig. 5.39. Effects of different saturation zones and vegetative treatment (with or without) on the soil moisture distribution in margin, slope and bottom from 16 September to 30 September 2014. Error bars indicate standard deviation from the mean. Means with the same letter do not differ significantly from each other.

A two-way ANOVA showed that the soil moisture distribution in October 2014 was significantly affected by different saturation zones (P<0.001) and vegetative treatment (with or without) (P<0.001) (Fig. 5.40). The interaction between saturation zones and the presence of vegetative treatment was significant (P<0.001). The bottom was the wettest zone with the greatest VWC mean value, while the margin was the driest with the lowest VWC. The non-vegetated gap had greater mean value in VWC in every saturation zone than that of the vegetated experimental unit.

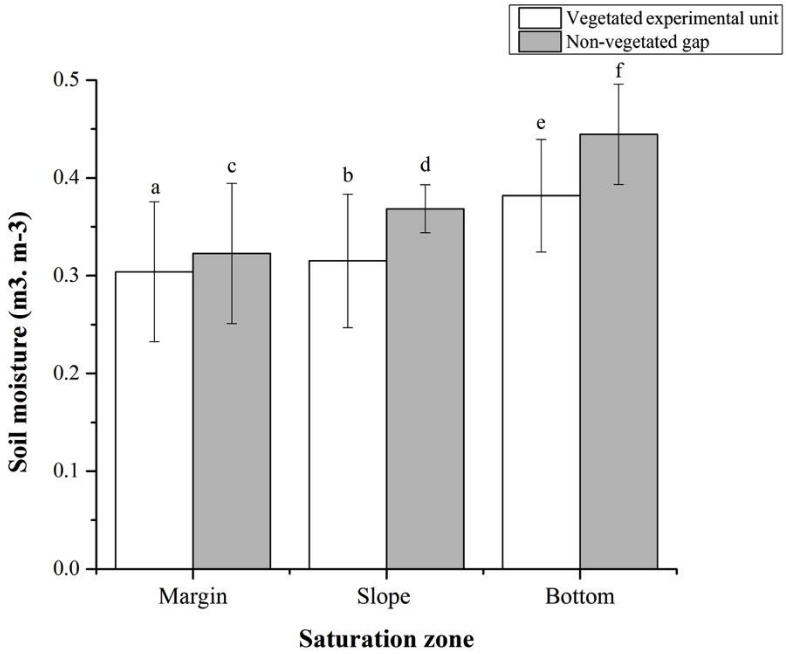


Fig. 5.40. Effects of different saturation zones and vegetative treatment (with or without) on the soil moisture distribution in margin, slope and bottom in October 2014. Error bars indicate standard deviation from the mean. Means with the same letter do not differ significantly from each other.

A two-way ANOVA showed that the soil moisture distribution in November 2014 was significantly affected by different saturation zones (P<0.001) and vegetative treatment (with or without) (P<0.001) (Fig. 5.41). The interaction between saturation zones and the presence of vegetative treatment was significant (P<0.001). Bottom was the wettest zone in both the non-vegetated gap and vegetated experimental unit. In the vegetated experimental unit the margin had the lowest mean value in VWC, while the slope had a VWC mean value that was lower than in the bottom but higher than in the margin. However, in the non-vegetated gap, the margin had a higher VWC mean value than in the slope. Generally, the non-vegetated gap had greater mean values in VWC from its margin and bottom than that of the vegetated experimental unit. However, the vegetated slope had a greater VWC mean value than in the non-vegetated slope.

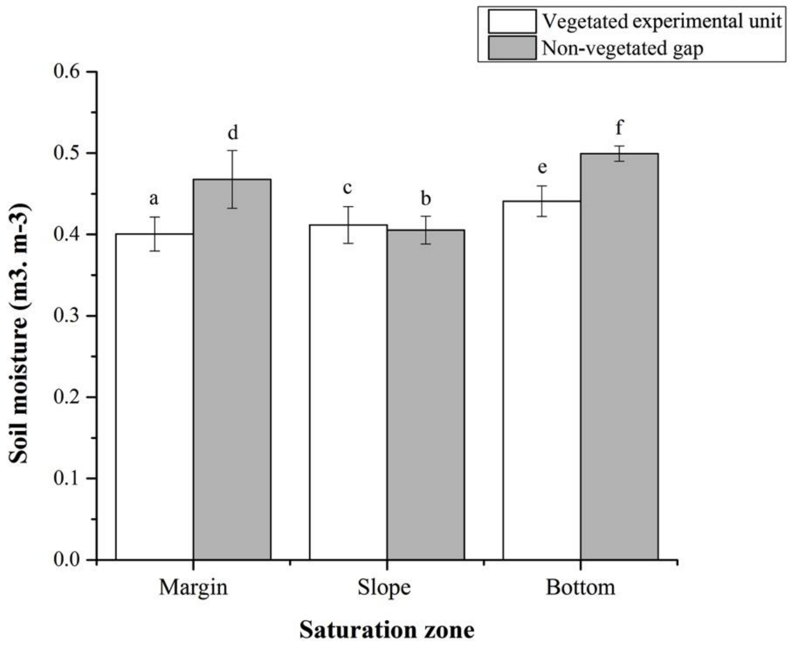


Fig. 5.41. Effects of different saturation zones and vegetative treatment (with or without) on the soil moisture distribution in margin, slope and bottom in November 2014. Error bars indicate standard deviation from the mean. Means with the same letter do not differ significantly from each other.

A two-way ANOVA showed that the soil moisture distribution from 1 December to 18 December 2014 was significantly affected by different saturation zones (P<0.001) and vegetative treatment (with or without) (P<0.001) (Fig. 5.42). The interaction between saturation zones and the presence of vegetative treatment was significant (P<0.001). In the vegetated experimental unit, the bottom was the wettest with the highest VWC mean value, while the margin had the lowest mean value in VWC. In the non-vegetated gap, the bottom was the wettest with the highest VWC mean value, while the slope was the driest zone with the lowest mean value in VWC. The non-vegetated bottom and margin had greater VWC mean values than in the vegetated bottom and margin. However, mean VWC value in the vegetated slope was higher than in the non-vegetated slope.

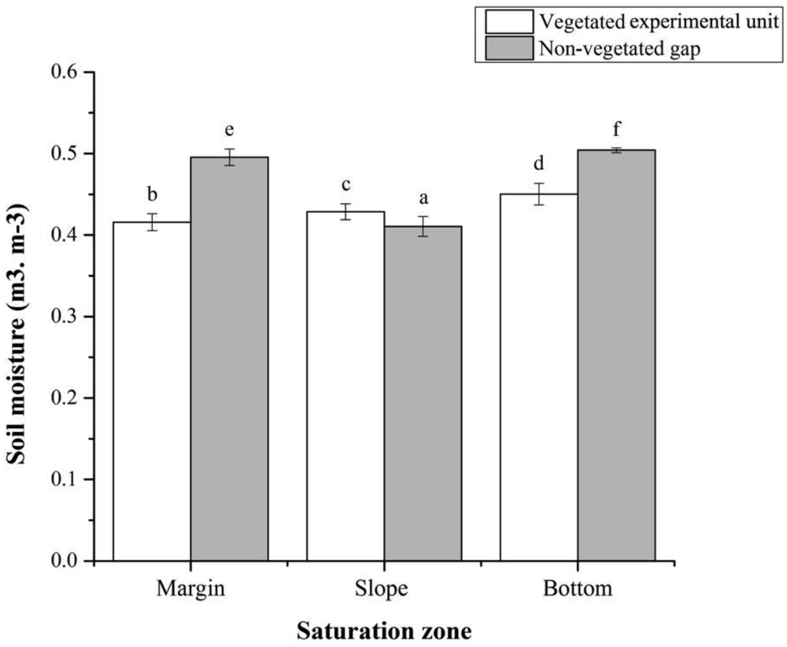


Fig. 5.42. Effects of different saturation zones and vegetative treatment (with or without) on the soil moisture distribution in margin, slope and bottom from 1 December to 18 December 2014. Error bars indicate standard deviation from the mean. Means with the same letter do not differ significantly from each other.

To sum up, generally, the non-vegetated gap had greater soil moisture than in the vegetated experimental unit. Johnston (2011) suggested that vegetation not only altered the soil structure to enhance the infiltration (i.e. the downward movement of soil water), but also improved the evapotranspiration, thus there would be a higher loss in soil moisture from the vegetated experimental unit than from the non-vegetated gap via infiltration and evapotranspiration. The depression bottom had the wettest saturation zone in both non-vegetated gap and vegetated experimental unit. The non-vegetated margin was drier than the non-vegetated slope in September and October 2014, however, the non-vegetated margin tended to be wetter than the non-vegetated slope from November to December 2014. This was partly affected by the increased on-site precipitation in November and December 2014 compared to that of September and October 2014, and partly due to the reduced evapotranspiration caused by the greatly decreased temperature in November and December 2014 compared to that of September and October 2014. The vegetated slope had constantly higher soil moisture than in the vegetated margin, which was expected.

5.2.5 Statistical analysis

The experimental design was a split-plot design which means mixed modelling. Generalised Linear Models for mixed models are relatively rare within software. This means that either the design element could be ignored, or data transformations for proportion data (i.e. % cover per quadrat) and counts of number of sown/unsown species should be involved. Using the wrong type of the variance-covariance matrix has far more serious consequences for the analysis than the distributional assumptions. Therefore the decision was taken to transform these stated variables prior to statistical analyses. The logit transformation was used for the % cover as it was proportion. It is a transformation that transfers a 0-100 scale onto a minus infinity to infinity scale with 50% being zero. It therefore adjusts for the hard ends of a percentage that has to fall between 0 and 100. The species data which were counts were transformed using a logarithm transformation as this is a good equivalence to the log-linear model which is how Poisson data is analysed using GLMs. This could also cope with both the study design and the nature of the data.

To analyse treatments (e.g. the use of felt mat and mulching) and saturation zone effects, vegetation community data for the logit transformed % cover per quadrat, the logged number of sown species and unsown ruderal species per quadrat, and the number of seedlings of sown species and unsown ruderal species per quadrat were analysed using General Linear Model (GLM) repeated measures in SPSS 20.0. This approach included treatment interactions with saturation zone, where the saturation zone was treated as a split plot factor within whole plots. In repeated measures ANOVA model for the tested datasets were checked using Levene's test for equality of variance, plots of Standard Deviation versus means and Normal Probability plots. There was no conclusive evidence that the assumptions were infringed and therefore the analysis was continued with. Greenhouse-Geisser correction was used to adjust F-statistic and P-values associated with main and interaction effects of saturation zone for possible violations of sphericity assumptions indicated by Mauchly's test. Transformed data are back transformed for presentation.

The response of the species composition to the treatments (i.e. the application of felt mat and mulching) and saturation zone was tested using Redundancy Analysis (RDA) in R's 'Vegan' package, following the methodology described by Oksanen (2015) and Oksanen et al. (2007). However, it is worth noting that the data from individual species does not satisfy the basic requirements for Principal Component Analysis (PCA). For instance, only 45 quadrats were accessed across the present experiment, however, it is good practice to have substantially more quadrats in a PCA analysis (Osborne & Costello, 2009), thus the basic assumption that species data is normal in all species does not hold. Further the split plot design of this work is ignored in carrying out a PCA. These problems are not solved by using more advanced techniques such as RDA, which actually involves a double PCA. However, there are some examples (e.g. Westbury & Dunnett, 2008) have used it on similar data sets, and have produced informative diagrams to draw conclusion from. The end result is that all use of these techniques can only be considered exploratory and non-robust.

The explanatory variable ‘saturation zone’ contains three classes: margin, slope and bottom. However, to avoid collinearity, the RDA model tends to process only two of the classes. Therefore, the set-up of the dataset defines two new terms including ‘edge’ and ‘position’. The term ‘edge’ represents margin and bottom. If an observation was from margin or bottom (i.e. the ‘edge’), edge was set to 1, and slope to 0. The same was done for slope. For the term ‘position’, the explanatory variables including margin, slope and bottom was coded with values 1, 2 and 3, respectively. For the three treatments including the application of felt mat, mulching 20 mm and mulching 100 mm, where the value is coded with 1 if sampling took place in the corresponding treatment and 0 elsewhere. As the relationship between the species and the species and explanatory variables are the primary interest in this experiment, the species conditional scaling was used. It would provide a triplot in which angles between species and explanatory variables could be interpreted in terms of correlations. Overall significance was accessed by Monte Carlo permutation tests under reduced model (999 permutations).

5.3 Results

5.3.1 % Cover

Results of GLM repeated measures were provided in Table 5.4. It indicates that both treatments (mulching and felt mat) had significant effects for % cover per quadrat of the established community (Table. 5.4). The use of fabric barriers significantly increased % cover per quadrat (Fig. 5.4). It appears that mulching with greater depth (100 mm) contributed to significantly greater % cover compared to the groups with less mulching depth (20 mm) and no mulching (Fig. 5.43). The interaction between mulching and felt mat was significant (Table. 5.4). In plots without application of felt mat, 100 mm mulching contributed to significantly increased % cover per quadrat compared with 20 mm mulching and bare soils that were not statistically different from each other (Fig. 5.43). In plots where felt mats were adopted, greater depth of mulching (100 mm) significantly increased % cover per quadrat compared with 20 mm mulching (Fig. 5.43).

Table 5.4. Determined effects of treatments and saturation zone for % cover per quadrat

	F	***P***
Mulching	38.300	<0.001 (***)
Felt mat	16.613	0.002 (**)
Mulching × Felt mat	7.876	0.019 (*)
Saturation zone	36.966	<0.001 (***)
Saturation zone × Mulching	3.217	0.034 (*)
Saturation zone × Felt mat	2.110	0.147 (ns)
Saturation zone × Mulching × Felt mat	0.126	0.883 (ns)

ns = not significant, *=between 0.05 and 0.01 **= between 0.01 and 0.001 and ***=<0.001

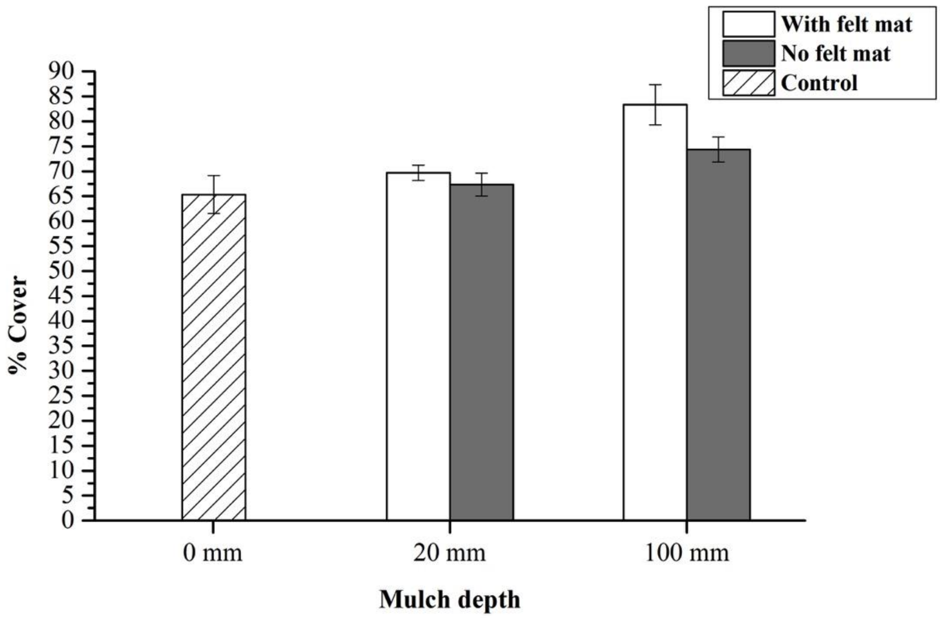


Fig. 5.43. Mean values of the % cover per quadrat, depending on treatments and irrespective of saturation zones. Error bars represent standard error.

Saturation zone was determined to significantly influence the community % cover per quadrat (Table. 5.4). Slope had the greatest % cover per quadrat amongst three saturation zones, while margin and bottom effects were not statistically significant (Fig. 5.4). Interaction between saturation zone and mulching was significant, whereas no interaction between saturation zone and felt mat was determined (Table 5.4). In margin and slope, plots applied 100 mm mulching contributed to greater % cover per quadrat than plots applied only 20 mm mulching and bare soils, which were found not statistically different (Fig. 5.43). In bottom, treatments adopting no mulching, 20 mm and 100 mm mulching showed no statistical differences in % cover per quadrat (Fig. 5.43). In bare soils, margin had the least % cover per quadrat than in slope and bottom, in which % cover per quadrat was found not statistically different from each other (Fig. 5.43). In plots applied 20 mm mulching, slope had the greatest % cover per quadrat, while % cover per quadrat in margin and bottom were not statistically different (Fig. 5.43). In plots applied 100 mm mulching, slope had the greatest % cover per quadrat, followed by margin, and bottom (Fig. 5.43). No interaction was found between the two treatments and saturation zone (Table 5.4).

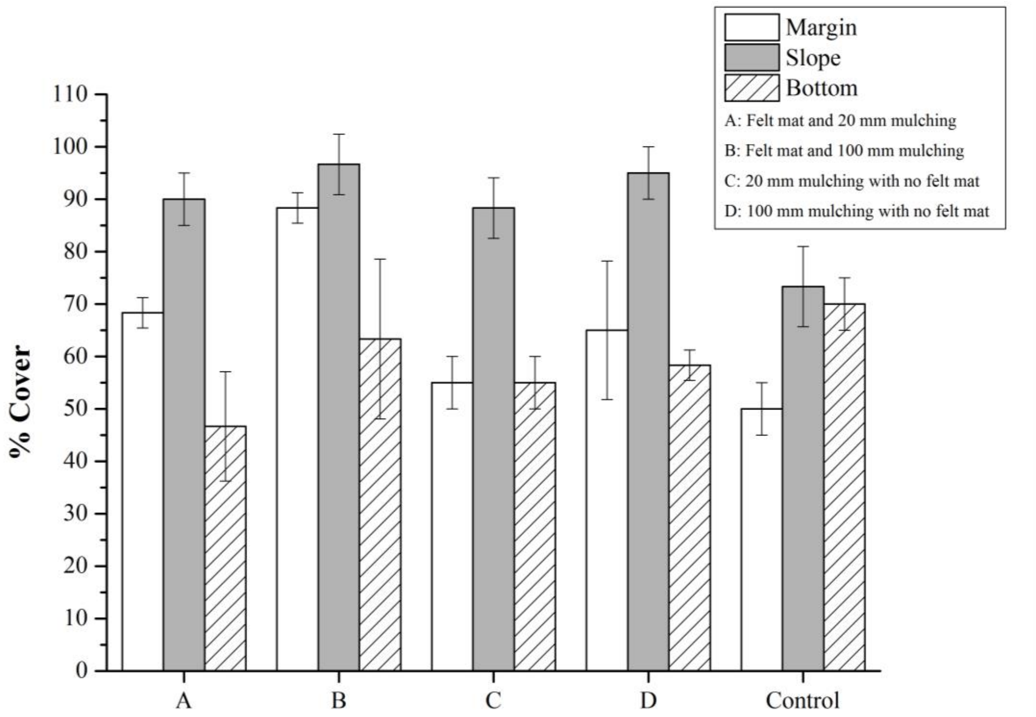


Fig. 5.44. Mean values of the % cover per quadrat, depending on treatments and saturation zones. Error bars represent standard error.

5.3.2 Number of sown species

Results of GLM repeated measures were provided in Table 5.5, which show that the effect of mulching was significant, whereas the effect of the application of felt mat was not statistically significant. It appears that plots applied mulching had significantly greater number of sown species per quadrat than in plots without application of mulching (Fig. 5.45). Nevertheless, plots applied different mulching depths (20 mm and 100 mm) showed no statistical differences in the number of sown species per quadrat (Fig. 5.45). The interaction between mulching and felt mat was not significant (Table 5.5).

Table 5.5. Determined effects of treatments and saturation zone for the number of sown species per quadrat

	F	*P*
Mulching	9.790	0.004 (**)
Felt mat	0.298	0.597 (ns)
Mulching × Felt mat	0.111	0.746 (ns)
Saturation zone	31.304	< 0.001 (***)
Saturation zone × Mulching	0.205	0.846 (ns)
Saturation zone × Felt mat	0.055	0.852 (ns)
Saturation zone × Mulching × Felt mat	1.402	0.267 (ns)

ns = not significant, *=between 0.05 and 0.01 **= between 0.01 and 0.001 and ***=<0.001

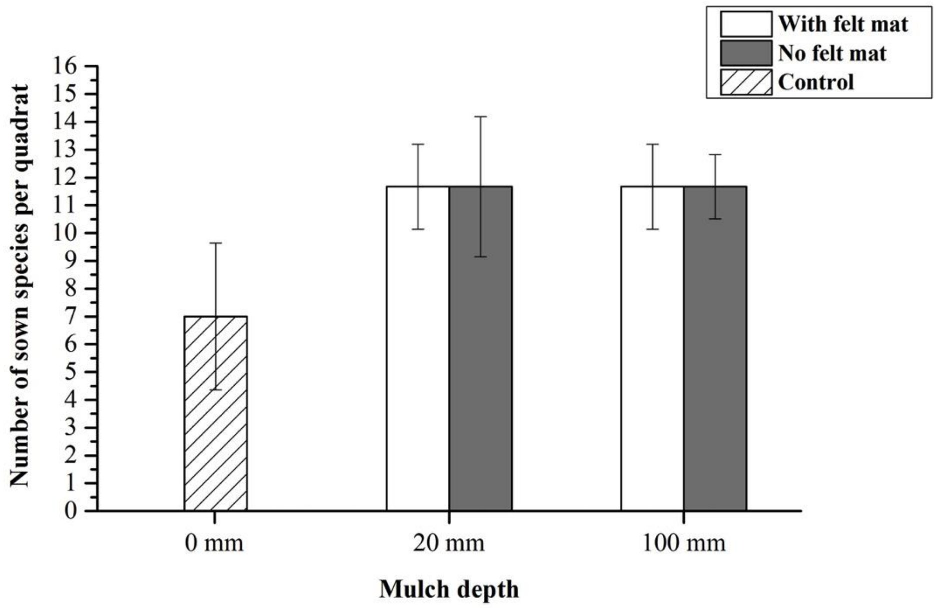


Fig. 5.45. Mean values of the number of sown species per quadrat, depending on treatments and irrespective of saturation zones. Error bars represent standard error.

Saturation zone was determined to significantly influence the number of sown species per quadrat (Table 5.5). Slope had the greatest number of sown species per quadrat, followed by margin, while bottom had the least number of sown species per quadrat (Fig. 5.46). Interaction between saturation zone and mulching, interaction between saturation zone and felt mat, as well as that of between saturation zone and mulching and felt mat were determined to be not statistically significant (Table 5.5).

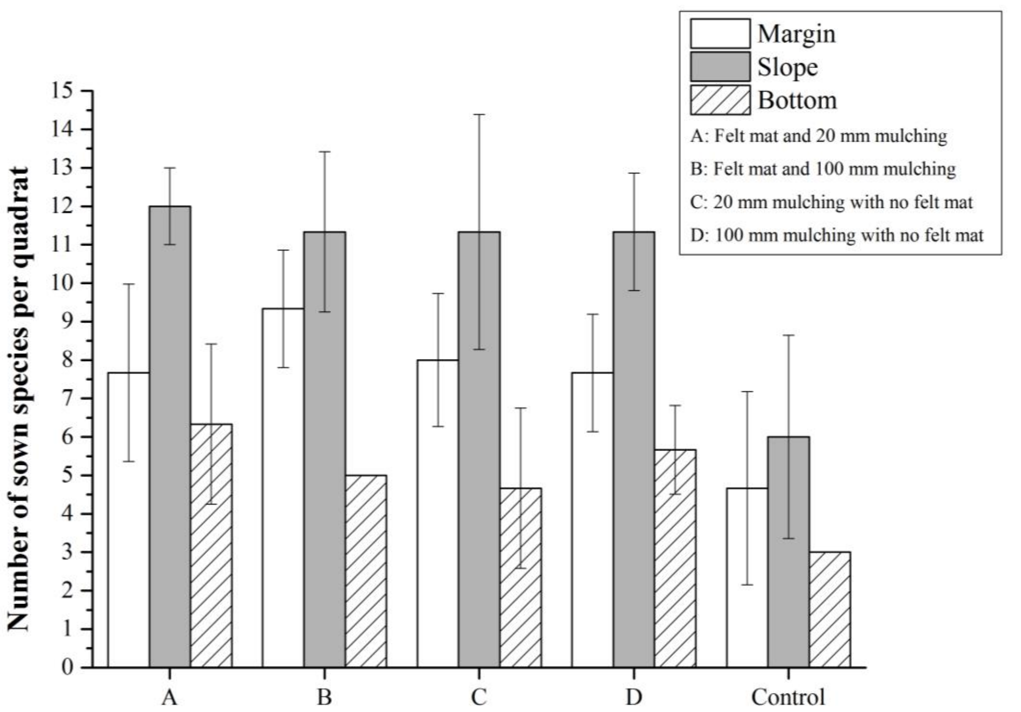


Fig. 5.46. Mean values of the number of sown species per quadrat, depending on treatments and saturation zones. Error bars represent standard error.

5.3.3 Number of unsown ruderal species

There were 13 ruderal forb species in total that occurred amongst the 15 experimental units during this study, which included: *Atriplex patula, Chamerion* *angustifolium, Galium aparine, Myosotis arvensis, Papaver rhoeas, Plantago lanceolata, Polygonum persicaria, Ranunculus repens, Rumex obtusifolius, Sonchus oleracea, Taraxacum officinale, Trifolium repens, Urtica dioica.* Table 5.6 shows the characteristics of the unsown ruderal species across the experiment, and their assumed hydrological regime and assumed distribution throughout the 'margin-slope-bottom' rain garden profile.

Table 5.6. Characteristics of the unsown ruderal species across the experiment and their assumed hydrological regimes and assumed distribution throughout the 'margin-slope-bottom' saturation zones in the rain garden. (Adapted from Hessayon, 2015; Lipper & Podlech, 1994; Schauer, 1982; Williams et al., 2003)

Species	Plant type	Characteristic	Assumed hydrological regime	Margin	Slope	Bottom
Atriplex patula	Forb	Annual weed of waste and arable land. Lobed or unlobed leaves but not separated into leaflets. Seeds ripen in August to October.	Infrequent inundation	*	*	
Chamerion angustifolium	Forb	Perennial, commonly occurs in open fields. Erect, smooth stem with scattered alternate leaves. Purplish-pinky flower from July to September. Seeds ripen in August to October.	Periodic or seasonal inundation	*	*	*
Galium aparine	Forb	Annual weed of arable land and waste places. Creeping straggling stems grow along the ground and over other plants. Narrowly oblanceolate leaves with hooked hairs. Seeds ripen in August to September.	Periodic or seasonal inundation	*	*	
Myosotis arvensis	Forb	Annual weed of cultivated fields and disturbed ground. Basal leaves with widely winged stalk. Curved cyme of small and blue-grey flowers occurs from spring to mid-autumn. Seeds ripen in July to September.	Periodic or seasonal inundation	*	*	*
Papaver rhoeas	Forb	Annual weed of arable land and waste places. Erect, hairy stem. Narrow leaves with toothed segments. Large flower in late spring. Seeds ripen in August to September.	Infrequent inundation	*	*	
Plantago lanceolata	Forb	Perennial. Rosette-forming with lanceolate leaves spreading or erect. Seeds ripen in July to September.	Periodic or seasonal inundation	*	*	
Polygonum persicaria	Forb	Perennial. Rosette-forming with lanceolate leaves spreading or erect. Seeds ripen in August to October.	Periodic or seasonal inundation	*	*	*
Ranunculus repens	Forb	Perennial weed with erect, hairless stem. Leaves deeply divided into three lobes or split into three leaflets. Small, shiny yellow flowers in spring to mid-summer. Seeds ripen in July to October.	Periodic or seasonal inundation	*	*	*
Rumex obtusifolius	Forb	Perennial of waste ground. Unbranched stems are often reddish. Large ovate leaves, lobed or unlobed but not separated into leaflets. Seeds ripen in July to October.	Periodic or seasonal inundation	*	*	

Sonchus oleracea	Forb	Annual weed of waste field and roadsides. Alternate leaves are deeply lobed with the leaf bases clasping the stem. Seeds ripen in July to September.	Infrequent inundation	*	*	
Taraxacum officinale	Forb	Perennial. Oblanceolate, oblong, or obovate leaves growing upright or horizontally spreading, with the bases gradually narrowing to the petioles. Seeds ripen in May and June.	Infrequent inundation	*	*	
Trifolium repens	Forb	Perennial. Creeping plants with trifoliate leaves of elliptic to obovate. Seeds ripen in July to October.	Periodic or seasonal inundation	*	*	*
Urtica dioica	Forb	Perennial. Erect stem with toothed, stinging hairy leaves. Seeds ripen in June to October.	Periodic or seasonal inundation	*	*	

* Assumed distribution

Neither treatment effects were determined to be significant, nor the interaction between mulching and felt mat (Table 5.7). Mean values of the number of unsown ruderal species per quadrat depending on treatments and irrespective of saturation zones are demonstrated in Fig. 5.47.

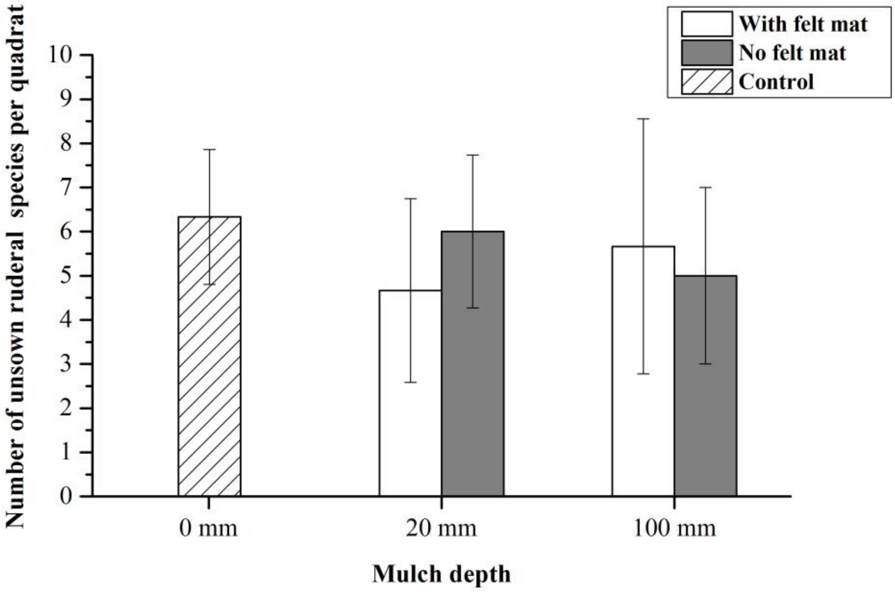


Fig. 5.47. Mean values of the number of unsown ruderal species per quadrat, depending on treatments and irrespective of saturation zones. Error bars represent standard error.

Table 5.7. Determined effects of treatments and saturation zone for the number of unsown ruderal species per quadrat

	F	P
Mulching	0.177	0.841 (ns)
Felt mat	0.174	0.685 (ns)
Mulching × Felt mat	0.923	0.359 (ns)
Saturation zone	15.289	< 0.001 (***)
Saturation zone × Mulching	0.993	0.434 (ns)
Saturation zone × Felt mat	0.341	0.715 (ns)
Saturation zone × Mulching × Felt mat	1.313	0.291 (ns)

ns = not significant, *=between 0.05 and 0.01 **= between 0.01 and 0.001 and ***=<0.001

Saturation zone was determined to significantly influence the number of unsown ruderal species per quadrat (Table. 5.7). It appears that bottom had the least number of unsown ruderal species per quadrat, while the effects of margin and slope were not statistically different from each other (Fig. 5.48).

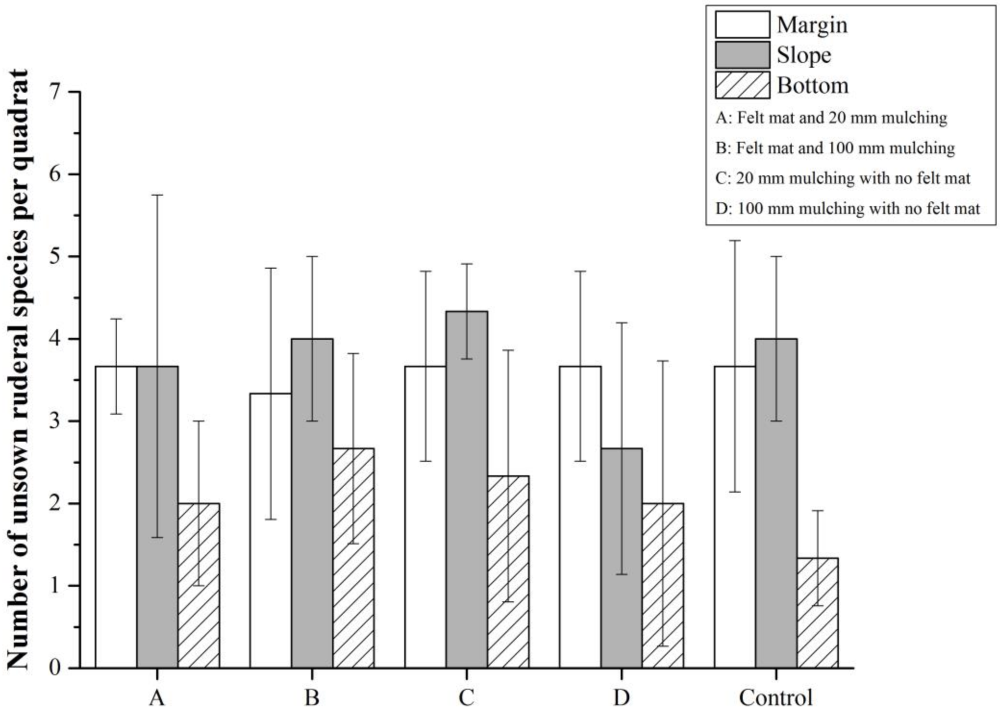


Fig. 5.48. Mean values of the number of unsown ruderal species per quadrat, depending on treatments and saturation zones. Error bars represent standard error.

5.3.4 Number of seedlings of sown species

Fig. 5.49 shows the mean values of the number of seedling of sown species per quadrat across the experiment. Results of GLM repeated measures were provided in Table 5.8. Results show that mulching had a significant influence on the number of seedlings of sown species per quadrat (Table 5.8). It appears that plots applied mulching had greater number of seedlings of sown species per quadrat than in the control group with bare soils (Fig. 5.49). However, effects of different mulching depths (20 mm and 100 mm) were not statistically different from each other (Fig. 5.49). Felt mat had no significant influence on the number of seedlings of sown species per quadrat (Table 5.49). No interaction between mulching and felt mat was found (Table

5.8).

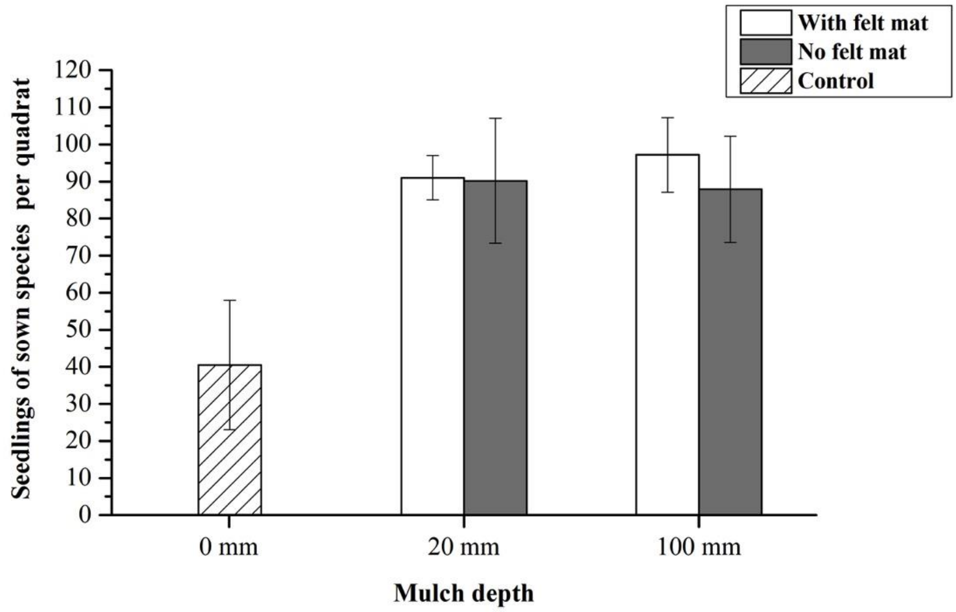


Fig. 5.49. Mean values of the number of seedlings of sown species per quadrat, depending on treatments and irrespective of saturation zones. Error bars represent standard error.

Table 5.8. Determined effects of treatments and saturation zone for the number of seedling of sown species per quadrat

	F	***P***
Mulching	16.914	0.001 (**)
Felt mat	0.424	0.529 (ns)
Mulching × Felt mat	0.281	0.607 (ns)
Saturation zone	153.367	<0.001 (***)
Saturation zone × Mulching	10.595	<0.001 (***)
Saturation zone × Felt mat	1.067	0.363 (ns)
Saturation zone × Mulching × Felt mat	0.43	0.656 (ns)

ns = not significant, *=between 0.05 and 0.01 **= between 0.01 and 0.001 and ***=<0.001

Saturation zone was determined to significantly influence the number of sown species per quadrat (Table. 5.8). Slope had the greatest number of seedlings of sown species per quadrat, while the effects of margin and bottom were not statistically different from each other (Fig. 5.50). Interaction between saturation zone and mulching was determined (Table 5.8). It appears that in margin and bottom, the numbers of seedlings of sown species per quadrat in plots applied 20 mm mulching, 100 mm mulching and those of bare soils were not statistically different from each other (Fig. 5.50). In slope, plots applied mulching had greater number of seedlings of sown species per quadrat than in bare soils (Fig. 5.50). However, the number of seedlings of sown species per quadrat from slope of the plots applied greater mulching depth (100 mm) showed no statistically significant difference from those applied shallower mulching (20 mm) (Fig. 5.50). In plots with bare soils, the number of seedlings of sown species per quadrat from margin was not statistically different from that of slope and bottom, however, slope had significantly greater number of seedlings of sown species per quadrat than in bottom (Fig. 5.50). In plots applied 20 mm and 100 mm mulching, slope had the greatest number of seedlings of sown species per quadrat, while the results from margin and bottom were not statistically different from each other (Fig. 5.50) Neither interaction between saturation zone and felt mat, nor the interaction between saturation zone, mulching and felt mat was suggested (Table 5.8).

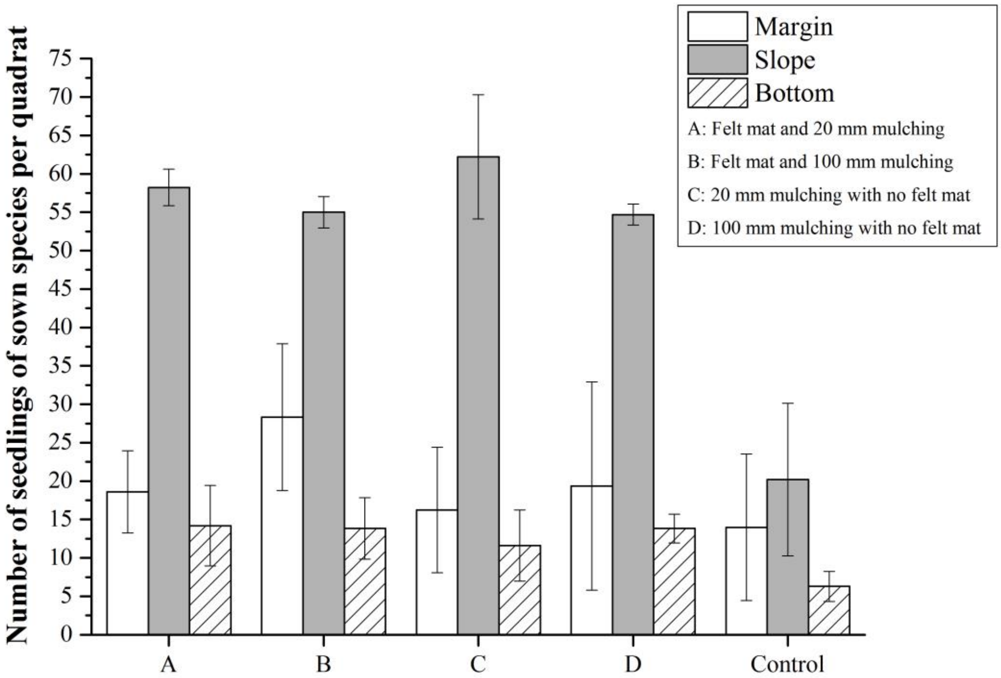


Fig. 5.50. Mean values of the number of seedlings of sown species per quadrat, depending on treatments and saturation zones. Error bars represent standard error.

5.3.5 Number of seedlings of unsown ruderal species

Results of GLM repeated measures were provided in Table 5.9. Mulching had a significant influence on the number of seedlings of unsown ruderal species per quadrat (Table 5.9). It appears that plots applied mulching had less number of seedlings of unsown ruderal species per quadrat than in the control group with bare soils (Fig. 5.51). However, effects of different mulching depths (20 mm and 100 mm) were not statistically different from each other (Fig. 5.51). Felt mat had no significant influence on the number of seedlings of unsown ruderal species per quadrat (Table 5.9). No interaction between mulching and felt mat was determined (Table 5.9).

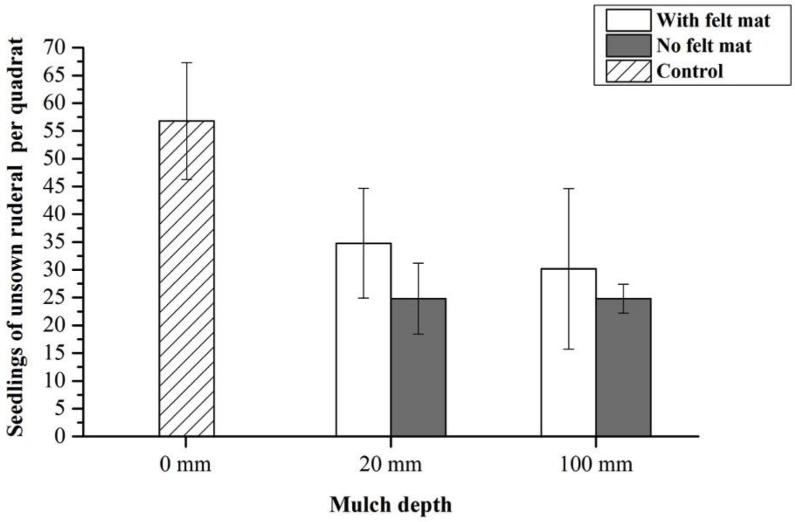


Fig. 5.51. Mean values of the number of seedlings of unsown ruderal species per quadrat, depending on treatments and irrespective of saturation zones. Error bars represent standard error.

Table 5.9. Determined effects of treatments and saturation zone for the number of seedling of unsown ruderal species per quadrat

	F	***P***
Mulching	10.324	0.004 (**)
Felt mat	1.905	0.198 (ns)
Mulching × Felt mat	0.17	0.689 (ns)
Saturation zone	3.43	0.052 (ns)
Saturation zone × Mulching	14.838	< 0.001 (***)
Saturation zone × Felt mat	0.197	0.823 (ns)
Saturation zone × Mulching × Felt mat	0.379	0.69 (ns)

ns = not significant, *=between 0.05 and 0.01 **= between 0.01 and 0.001 and ***=<0.001

Saturation zone had no significant influence on the number of seedlings of unsown ruderal species per quadrat (Table 5.9). However, interaction between saturation zone and mulching was suggested (Table 5.9). Mulching had no influence on the number of seedlings of unsown ruderal species per quadrat in margin and bottom (Fig. 5.52). In slope, control group with bare soils had significantly greater number of unsown ruderal species per quadrat than in the plots applied mulching (Fig. 5.52). However, the number of unsown ruderal species per quadrat from slope in plots applied 20 mm and 100 mm mulching had no significantly statistical difference between each other (Fig. 5.52). In plots applied mulching, number of unsown ruderal species per quadrat from different saturation zone was not statistically different from each other (Fig. 5.52). In control group with bare soils, slope had the greatest number of unsown ruderal species per quadrat, while the results from margin and bottom were not statistically different (Fig. 5.52).

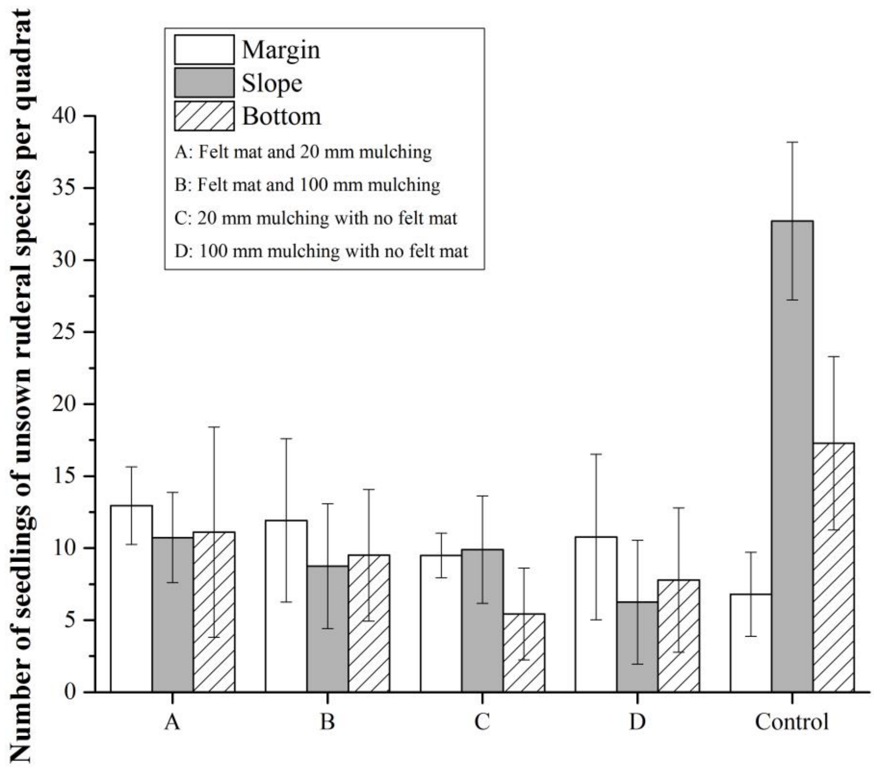


Fig. 5.52. Mean values of the number of seedlings of unsown ruderal species per quadrat, depending on treatments and saturation zones. Error bars represent standard error.

5.3.6 Species composition and performances of individual species

On 19 September 2014, 45 quadrats (each quadrat was calculated as the mean of three replicates) were recorded. Presence of individual species across the experiment, irrespective of treatments (felt mat and mulching) and saturation zone is shown in Fig. 5.53. In Fig. 5.53, species organised themselves at the right of the line in the centre of the figure had germinations in more than half of the total quadrats, and were therefore considered as common species in community. In September 2014, *Achillea millefolium*, *Deschampsia cespitosa* and *Leucanthemum vulgare* were the dominant sown species that grew in most quadrats (42, 45 and 45 out of the total 45 quadrats, respectively). *Chamerion angustifolium* was the unsown ruderal dominant that germinated in 44 out of 45 quadrats (Fig. 5.53). The sown *Achillea millefolium* and *Leucanthemum vulgare*, as well as the unsown ruderal *Chamerion angustifolium* had the greatest number of seedlings per quadrat (8.01, 7.61 and 6.33, respectively) excluded quadrats where absent, whereas the other sown/unsown ruderal species had less than 5 seedlings per quadrat (Fig. 5.53).

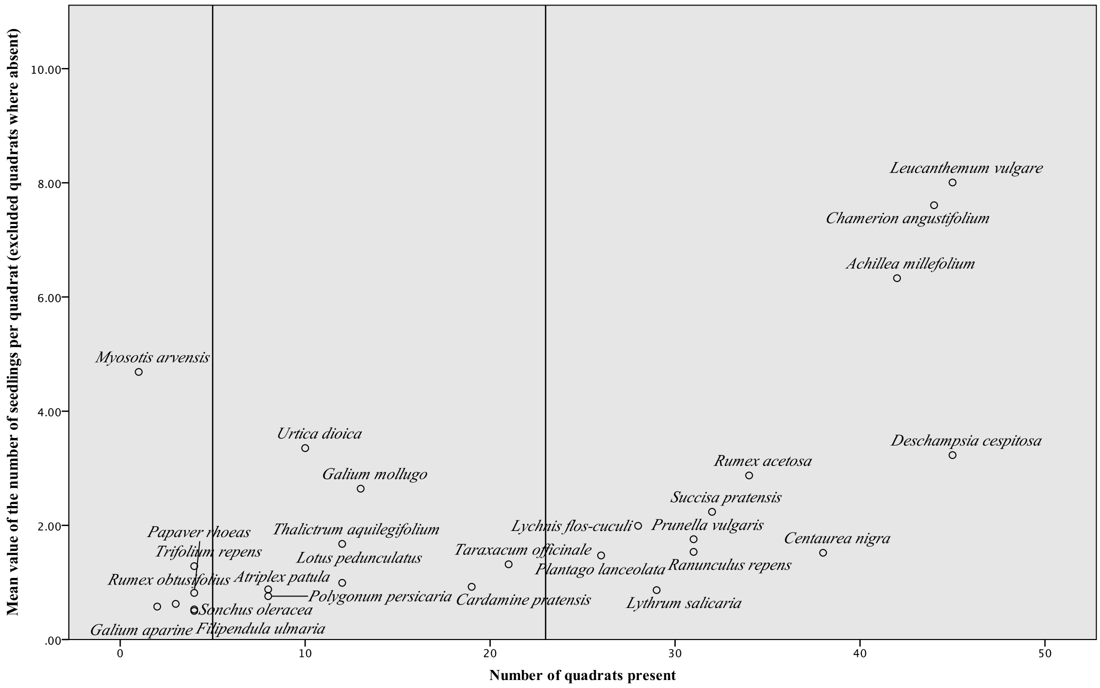


Fig. 5.53. Species presence across the experiment in September 2014, irrespective of treatments and saturation zone

In Fig. 5.53, there are 14 species organised themselves at the left of the centre line had emergence in less than half of the total quadrats, which are therefore considered as rare species in the experimental trials. *Cardamine pratensis, Filipendula ulmaria, Lotus pedunculatus* and *Thalictrum aquilegifolium* were the only four rare species from the sown seed mixes, while most of the rare species were unsown ruderal species. It is noticeable that the sown *Filipendula ulmaria*, as well as six unsown ruderal species including *Galium aparine, Myosotis arvensis, Papaver rhoeas, Rumex obtusifolius, Sonchus oleracea* and *Trifolium repens* had remarkably limited emergences across the experiment, which were found in less than 5 out of 45 quadrats (Fig. 5.54). Although rain garden slope tended to have more presence of these particular rare species, it appears that the spatial distributions of them were rather random across the trials (Fig. 5.54). It is also worth noting that *Valeriana officinalis* had no emergence across the experiment in September 2014.

	A	**B**	**C**	**D**	**Control**
Margin		PR (1)[a]	GA (1) TR (1)		FU (1) GA (1)
Slope		PR (1) RO (1) SO (1)	FU (1) RO (1) PR (2)	FU (1)	MA (1) TR (1)
Bottom	MA (1)		FU (1)	MA (1)	

a: Number in parenthesis indicates the number of quadrats that had the specific

Abbreviations of species:
FU: *Filipendula ulmaria*
GA: *Galium aparine*
MA: *Myosotis arvensis*
PR: *Papaver rhoeas*
RO: *Rumex obtusifolius*
SO: *Sonchus oleracea*
TR: *Trifolium repens*

Treatment:
A: Felt mat + Mulching 20 mm
B: Felt mat + Mulching 100 mm
C: Mulching 20 mm
D: Mulching 100 mm
Control: No Treatment

Fig. 5.54. Presence of seven very rare species (*Filipendula ulmaria*, *Galium aparine*, *Myosotis arvensis*, *Papaver rhoeas*, *Rumex obtusifolius*, *Sonchus oleracea* and *Trifolium repens*) across the experiment

The resulting triplot is presented in Fig. 5.55. Numerical output suggests that all five explanatory variables (e.g. felt mat, mulching 20 mm, mulching 100 mm, position and edge) explain 51.75% of the variation in the species data. The two-dimensional approximation in Fig. 5.55 explained 78.39% of this (63.36% on axis 1 and 15.03% on axis 2). Therefore, the first two axes explain 40.57% of the total variation in the species data.

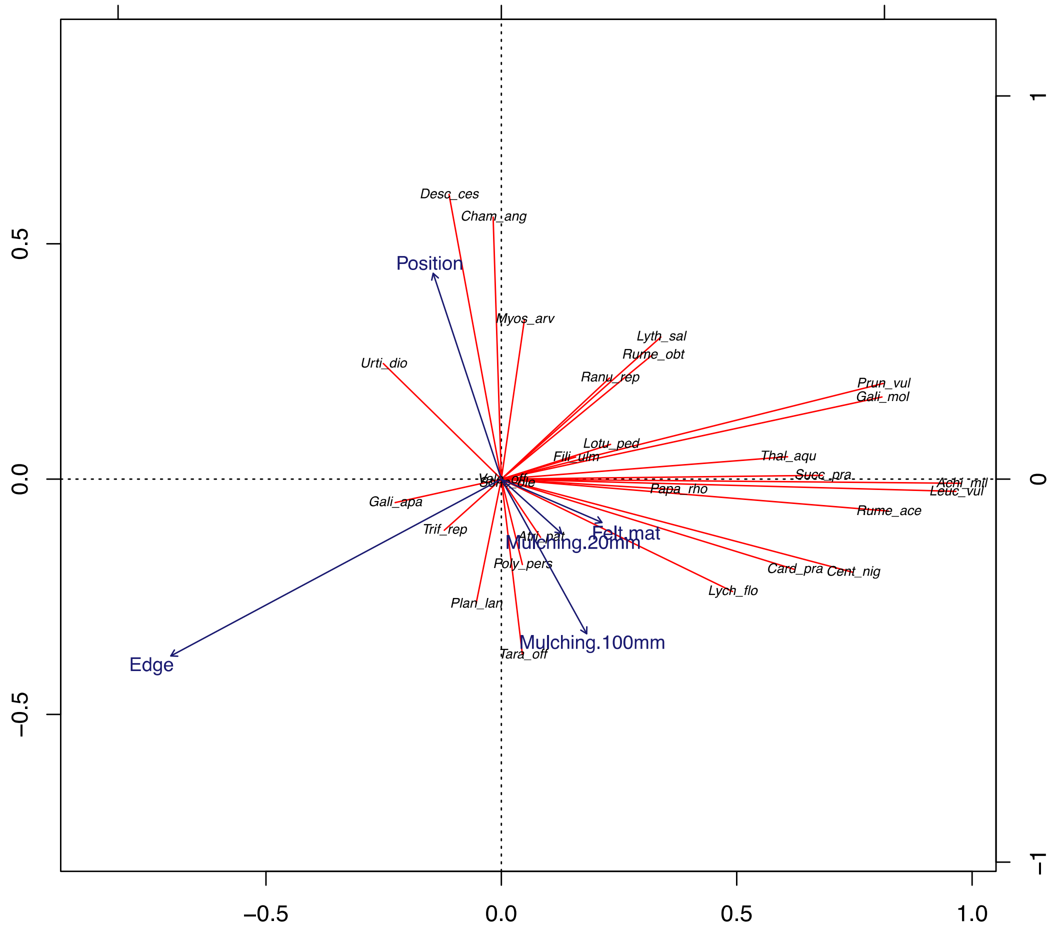


Fig. 5.55. Resulting triplot obtained by RDA

The results of permutation tests for RDA under reduced model were presented in Table 5.10. It indicates that felt mat effect is not significant on species emergence, whereas both mulching level is significantly related to the species data. There is a strong ‘edge’ effect demonstrating the plant emergences on slope were significantly independent from that of margin and bottom. Further the non-significant ‘position’ effect indicates that the species data from margin and bottom was not statistically different.

Table 5.10. *F*-statistic and *p*-values of conditional effects obtained by Monte Carlo permutation tests in RDA (999 permutations).

Explanatory variable	***F***	***P***
Felt mat	0.904	0.380 (ns)
Mulching 20 mm	9.994	0.005 (**)
Mulching 100 mm	11.762	0.005 (**)
Edge	25.644	0.005 (**)
Position	2.084	0.090 (ns)

ns= not significant, *=between 0.05 and 0.01 **= between 0.01 and 0.001 and ***=<0.001

Observations on the correlations between species and treatments and saturation zone derived from Fig. 5.56 are listed below,

(1) No valid conclusions could be drawn for the sown *Valeriana officinalis* (no emergence across the experiment), as well as the ruderal *Sonchus oleracea* due to rather little germination at the time of measurement.

(2) Most desirable sown species except *Deschampsia cespitosa* and *Lythrum salicaria* were positively responsive to the application of 100 mm mulching. Decreases in abundance due to the application of 100 mm mulching were found in 6 unsown ruderal species, including the dominant ruderal *Chamerion angustifolium*, with *Galium aparine, Myosotis arvensis,*

Ranunculus repens, *Rumex obtusifolius* and *Urtica dioica*. Although the remaining ruderal species were supported by the use of 100 mm compost mulching, their abundances responsive to mulching 100 mm were less than most of the sown species.

(3) Generally, species' responses to the application of 20 mm mulching were close to the results showed in the application of 100 mm mulching. *Lythrum salicaria* and the ruderal *Ranunculus repens* and *Rumex obtusifolius* were associated with in the application of 20 mm mulching, whereas the abundance of *Trifolium repens* was reduced due to the application of 20 mm mulching. However, it is noticeable that the effect of the application of 20 mm mulching on the four species was weak. It is also worth noting that *Ranunculus repens, Rumex obtusifolius* and *Trifolium repens* were rare species based on the decisions stated previously. The similarity between the results from the application of 20 mm and 100 mm mulching could also confirm the decision that the difference of mulching depths had no significant effect on the community data.

(4) All desirable sown species and most of the unsown ruderal species were positively responsive to slope. Only 6 unsown ruderal species tended to prefer margin and bottom, which include *Galium aparine*, *Plantago lanceolata*, *Polygonum persicaria*, *Taraxacum officinale*, *Trifolium repens*

and *Urtica dioica*.

(5) It is worth to emphasise that due to the nature of the RDA model, the 'position' effect can only explain whether if a specific species preferred to organise itself towards the drier marginal areas or the wetter bottoms. It is not possible to simply make decisions for the very rare species showing random distribution across the trials depending on the RDA triplot. We also consider that the species with data taken at around 90 ° angle to the 'position' arrow experienced rather weak 'position' effects and would show no statistical differences in abundances between margin and bottom, which could be applied in most sown and unsown ruderal species found across the experiment. A few species showed intolerance of the limited moisture conditions in rain garden margins but preferred the damp bottoms, including the sown *Deschampsia cespitosa* and the ruderal *Chamerion angustifolium, Myosotis arvensis* and *Urtica dioica.* In contrast, the sown *Lychnis flos-cuculi* and the unsown ruderal *Atriplex patula, Plantago lanceolata, Polygonum persicaria* and *Taraxacum officinale* tended to have more germinations in the marginal areas compared to bottoms.

Valid interpretation on the between-species relations could also be drawn from this RDA analysis. Generally, species excluding *Sonchus oleracea* and *Valeriana officinalis* could be divided into four components. Species within each component were often found to grow together.

Component 1 has 6 sown species (*Achillea millefolium, Cardamine pratensis, Centaurea nigra, Leucanthemum vulgare, Lychnis flos-cuculi,* and *Rumex acetosa*), and 4 unsown ruderal species (*Atriplex patula, Papaver rhoea, Polygonum persicaria* and *Taraxacum officinale*). Generally, species from this group were responsive to the two mulching treatments. Most of these species tended to have more emergences in slope rather than margin and bottom, while *Polygonum persicaria* and *Taraxacum officinale* showed more germination in margin and bottom.

Component 2 has three species including the only sown grass *Deschampsia cespitosa*, the dominant ruderal *Chamerion angustifolium* and a rare ruderal *Urtica dioica*. Only *Urtica dioica* had more germination in margin and bottom than in slope. This group has a negative correlation with species from component 1, which means these species will compete with species from component 1, and therefore they could not co-exist in this community. It appears that these species were suppressed by the application of mulching, and therefore the population of component 1 was promoted with the use of mulching.

Component 3 has 7 sown species including *Filipendula ulmaria, Galium mollugo, Lotus pedunculatus, Lythrum salicaria, Prunella vulgaris, Succisa pratensis* and *Thalictrum aquilegifolium,* and 3 unsown ruderal species including *Myosotis arvensis, Ranunculus repens* and *Rumex obtusifolius*. All species from component 3 showed more emergences in slope than in margin and bottom. Most of these species, except *Myosotis arvensis,* preferred the application of 20 mm mulching. *Lythrum salicaria* and three ruderal species including *Myosotis arvensis, Ranunculus repens* and *Rumex obtusifolius* showed decreased abundance due to the treatment with greater mulching depth to 100 mm. These species also showed positive correlation between species from component 2. It appears that all the sown species in component 3 were positively correlated with the sown species from component 1. Most sown species excluding *Lythrum salicaria* from component 3 showed either no correlation or negative correlation between species from component 2.

Component 4 has only three unsown ruderal species, which are also considered as rare species (*Plantago lanceolata*) or very rare species showed random distribution across the trials (*Galium aparine* and *Trifolium repens*). Species from this component had more germination in margin and bottom than in slope, and were negatively correlated with species from component 3. *Plantago lanceolata* were positively correlated with species from component 1, but were negatively correlated with species from component 2.

5.4 Discussion

5.4.1 Success of the experimental seeding rain gardens

In general, seeding as an alternative planting solution for rain garden showed relatively good results in this study. Planting coverage observed within the whole plots (i.e. weed control treatments) were considered good. The least % cover per quadrat was found in the control group (65.33%), whilst the treatment adopting felt mats and mulching to 100 mm showed the greatest % cover per quadrat (83.33%). Although the runoff retention and detention was not formally assessed, the dense coverage of the sown vegetation is expected to greatly increase the rainfall interception and evapotranspiration on-site, which may thus maintain the primary rain garden function of runoff quantitative control during the growing season.

During the experimental period, the control group had 7 out of 15 species germinated from the previously weeded ground, while 40.48 seedlings per quadrat was counted on-site. It is considered acceptable for a practice with extreme low funding and maintenance inputs. Furthermore, species richness (i.e. total number of different species per quadrat) and seedling density (i.e. seedlings of different species per quadrat) of the sown species increased considerably due to the adoption of the simple weed control approaches. All plots adopting weed control means had 11.67 out of 15 species germinated on-site, and their seedling densities were more than two-fold of that of the control group. As stated previously, planting diversity and density are two key factors governing the runoff retention and detention in rain gardens, which are also vital to the sustainability of other basic ecosystem services in the established community such as the interaction with local biodiversity (Fig. 5.56) and the seasonal display, as well as visual impact (Fig. 5.12). Therefore, we regard seeding rain garden as a viable solution for installation.

Fig. 5.56. Experimental seeding rain gardens visited by local pollinators (Photos were taken by the author in August 2014)

5.4.2 Effectiveness of the weed control means on the vegetation pattern of the sown forb-rich mixes

Results show that the use of felt mat and mulching increased % cover of the plant community. Plots adopting the combination of felt mat and compost mulching show significantly increased % cover per quadrat compared to those only adopted single treatment. Mulching showed a stronger effect on altering the vegetation coverage than the presence of the felt mat, while increased mulching depth positively contributed to the increase of % cover. Although greater planting coverage may promote runoff interception and loss through evapotranspiration, it is worth noting that the observed planting coverage included all germinated sown species and undesirable ruderal species on site. The assessed % cover data could not separate the sown species from that of the unsown ruderals. Therefore, only considering the between-group comparison of % cover is not sufficient for determining whether if the two involved weed control means had significant effects on the establishment of the sown community.

Species richness and seedling density of the desirable sown species were greatly increased in plots adopted compost mulching compared to the control group with bare soils. Plots adopted mulching also showed significantly less number of seedlings of the ruderal weeds per quadrat compared to the control group. The experimental outcomes match the initial expectation that the compost mulching could provide a head start for the emergence of the sown species to establish predominance in communities. However, clear evidence showed that greater mulch depth had no significant effects on the species richness and seedling density in the sown and the unsown ruderal species. In fact, the increased % cover of vegetation due to the application of greater mulching depth indicates that the nutrient-enriched situation may also possibly promote the growth of vigorous colonists over time. Sown plants would simply suffer competitive displacement by vigorous neighbours found in the productive soils, which may thus considerably reduce the functional diversity and ecosystem services derived from the desirable sown species. Moreover, since the green waste composts intrinsically have high potassium, nitrate and phosphate levels, excessive adoption of compost mulching in urban rain gardens may thus result in unfavorable leach of elements-polluted excess runoff to contaminate the connected water courses (Li et al., 1997). We therefore suggest that the future application should not adopt too much mulching to further reduce the installation costs, and to minimise the potential of harms of weed productivity and water eutrophication. It might worth to use a lower nutrient substrate for mulching in the future application. However, inorganic particulate substances such as sands and gravels shall not be used solely to form the seeding beds, because these materials could not retain sufficient moisture to aid germination and are incapable of stopping slope erosion.

The seed mixes were sown in December 2013, when there would be rather limited wind-born seeds to contaminate the treatment plots. Felt mats were expected to work as barriers to minimise the germination of ruderal species that spontaneously grew from the existing soil seed bank. Surprisingly, experimental outcomes may lead to a bias that the use of fabric barriers as a weeding solution failed to stop the emergence of ruderal seedlings. Considering the facts that the felt mats have no effect on suppressing the growth of the wind-born seeds and there were large number of ruderal weeds germinated on-site, it is thus reasonable to suspect that the stated "failure" of felt mats was probably resulted from the unreliability of the green waste compost. The used compost was freshly made and determined to be weed-free in mid-October 2013. However, the compost was left outdoors for two months until being used in the study site in mid-December 2013. During October and November, all the seeds of the existing ruderal species were ripened, so that the used compost might acquire significant load of wind-born seeds. If fabric barriers were virtually impossible to supress the spontaneous ruderal species that grew from the existing soil seed bank, then the plots adopted felt mats would have exactly the same number of ruderal species as that of the control group. However, plots adopted felt mats showed less number of ruderal species per quadrat compared to the control group, which reinforces the suspicion that the used compost mulching carried a significant load of wind-born seeds with viability and ability to contaminate the sown communities. Thus, we could hardly draw a valid conclusion for the effects of felt mats on weed control, and also suggest that the use of sterile compost mulching should be insured on the time of the mulching application in future study.

5.4.3 Effects of the 'margin-slope-bottom' saturation zones in practical rain gardens on the vegetation pattern of the sown forb-rich mixes

Overall, we consider the sown plantings were good practice which showed satisfying establishment in all three saturation zones. Experimental observation shows the spatiotemporal distribution of soil moisture resulted in a significant 'dry-wet' moisture gradient throughout the 'margin-slope-bottom' rain garden profile, which is initially assumed to greatly influence the vegetation pattern of the sown forb-rich mixes. Our initial expectation has been proved by experimental outcomes. Saturation zone showed significant effects on altering the planting coverage, species richness of the sown and unsown ruderal species, as well as the seedling density of the desirable sown species. The level of statistical significances for the determined effects of saturation zone were greater than that of weeding treatments, which clearly suggests that the moisture gradient throughout the rain garden soil profile was the most important factor in relation to the vegetation pattern of seeding rain gardens. It is noticeable that saturation zone showed no impacts on the seedling density of ruderal weeds. This again led to the suspicion that the adopted compost mulching was contaminated, in which the wind-born seeds were supposed to randomly germinate on-site and thus the saturation zone effects become insignificant.

Analysis for all within-subject effects in repeated measures ANOVA suggests that, in most cases, slopes in the vegetated experimental units had significantly higher % cover, species richness and seedling density of the sown species and the undesirable unsown ruderals than in the margins and bottoms across the experiment. Rain garden bottoms had the least number of sown and unsown ruderal species per quadrat, while the % cover per quadrat and the seedling density of sown species in the bottoms and margins were not statistically different. In this study, slopes with the moderate soil moisture is functioning as the ecotone between the frequently waterlogged bottoms and the drier marginal areas, which is richer in sown species than that of the margin and bottom. Considering the experimental observation and the fact that ecotone is often biologically richer than either habitat adjacent to them on either side (Goebel et al., 2003; Palmer & Mazzotti, 2004), we therefore consider the rain garden slope as the key area for promoting species diversity and structural diversity for the sown community and strongly recommend mixing more desirable species in seed mixes tended to be adopted for rain garden slope in future application. The limited species diversity and seedling density observed in rain garden bottoms and margins is probably a result of the wet/dry moisture-induced stress in plants. Although the seeding beds never got truly anaerobism during the experimental period, it is likely that the few big winter rainfall could wash some of the sown seeds away to reduce the seedling density. Mazer et al. (2001) suggested that repeated inundation occurring in bioswales (with the similar depression structure as the rain gardens) could severely limit germination of the sown seed mixes. There are also a few long dry spells between recorded precipitations during the experimental periods, which may greatly challenge the maintenance of the available soil moisture in marginal area. Stress resulting from the limited moisture availability during seed development is also considered constraint to reduce vegetation yields (Vieira et al., 1992).

In this study, saturation zone is a split-plot factor within the whole plots, interaction between the saturation zone and weeding treatments may provide more insights. In general, in most cases, mulching showed no effects on altering the establishment of sown rain garden mixes in the dry marginal areas and the wet depression bottoms. It appears that vegetation pattern in rain garden slope is most responsive to the application of compost mulching. For instance, the mulched slopes had significantly greater seedling density of sown species and less seedling density of ruderal weeds than in the control group adopted no mulching. It is understandable that the mulching provided a head start for the emergence of sown species to establish their advantages in early competition in the moist slopes. We also assume the mulched compost can keep the in-situ sown seeds at where they were originally in the seeding beds, whereas a large amount of seeds that were spread on the bare soils could be flushed away by the downstream runoff in rainy weather.

5.4.4 Performance of individual species across the experiment

In September 2014, the *Achillea millefolium, Deschampsia cespitosa* and *Leucanthemum vulgare* dominated the sown communities, and are therefore regarded as the keystone species in the first growing season. These species not only showed the greatest lab germination rate amongst all sown species, but also have been suggested to have the moisture preferences to tolerate a fairly wide range of soil moisture condition (Brickell, 2008). It is noticeable that *Leucanthemum vulgare* preferring in moist but well-drained soil (Brickell, 2008) was found well adapted to rain gardens even at the base of depression. This is not surprising considering its great lab germination rate and the fact that the rain garden bases never encountered truly anaerobism during the experimental period. Other sown species had much less abundances compared to the three dominants, and are thus regarded as "complementary" species in the sown community. Only four sown species including *Cardamine pratensis, Filipendula ulmaria, Lotus pedunculatus* and *Thalictrum aquilegifolium* were determined as the rare species with considerable absence across the experiment. This outcome is reasonable as the used seeds of *Cardamine pratensis, Filipendula ulmaria* and *Lotus pedunculatus* showed rather poor germination in relatively ideal laboratory conditions (40%, 4% and 45%, respectively), so that the poor field germination in the three species are expected. Although *Thalictrum aquilegifolium* showed good lab germination (82%), this non-robust species is considered incapable of developing competitive advantage in seed mixes. In September 2014, the keystone structure of the sown community was largely dominated by the three dominant keystone species, which provided sufficient canopy coverage and significant flower heads to maintain the community structure and colour display. The co-occurring species adopted the special foliage shapes and different structures, heights and spreads to maintain a relatively complex structural diversity. The keystone structure is assumed to be capable of maintaining the rainfall interception and evapotranspiration. As stated previously, the visual appearance of the keystone structure is considered pleasant.

The experimental behaviour of individual species showed that the felt mat effect is not significant on the species emergence. Abundances of most species, especially of those species showed fairly good emergences across the experiment (i.e. those are not determined as the rare species) were responsive to mulching treatment. Most of the sown species had significantly more germination in plots adopted mulching, while most of the ruderal species had decreased number of seedlings due to mulching. Effects of different mulching depths appear to be non-significant. Overall, evaluation of individual species would conduct exactly the same conclusions as which derived from the community data. Moreover, we found 9 out of 13 unsown ruderal species were rare species in the communities which had remarkably limited emergences across the experiment. *Chamerion angustifolium* was the only ruderal species showed a significant load of germination in the experimental units, and may consider a potential threat to community dynamics. However, the emergence of this species was largely reduced in plots adopted weeding treatments. All the stated experimental evidences again demonstrate the positive effects of the adopted simple weeding treatments on the establishment of the sown mixes, especially of the compost mulching. It is also worth noting that the abundance of *Deschampsia cespitosa* was negatively related to mulching. We therefore guess the greater emergences of the vigorous and fast-growing sown forbs (e.g. *Achillea millefolium* and *Leucanthemum vulgare*) contributed by the application of compost mulching would outcompete the sown grasses. However, we would not worry about such effect as grass species may usually develop their competitive advantages over time to threat the sown community dynamics. We thus consider the mulching effect is satisfied.

The emerged species from among the many sown species are likely to be those best adapted to the specific site conditions (Druse & Roach, 1994). In this study, distributions of most sown species found across the experiment match their preferred moisture content regimes and the assumed distribution positions throughout the “wet-dry” moisture gradient, in which the rain garden slopes were much preferred by all the sown species. The results match the conclusion derived from the community data, which again indicate that the moderately moist rain garden slopes could be prefect spots for the sown plantings to yield diverse and functional communities. However, it is worth noting that greater seedling density of the unsown ruderals was found in slope compared to that from margin and bottom. 7 out of 13 unsown ruderal species including the dominant *Chamerion angustifolium* preferred to locate themselves in rain garden slopes. Although the ruderal competitors did not develop competitive advantages during the experimental period, we may have to assume that the weed competition in rain garden slope would be severer than in margin and bottom through time. Therefore, further weeding treatments may be required for the rain garden slopes to sustain consistent performances.

Abundances of all the sown species were significantly reduced in rain garden margins and bottoms compared to that from the slopes. As previously stated, the repeated multi-day inundations in bottoms during wet weathers and the limited available moisture content in marginal areas could be the potential dominant factor to severely limit the seed germination and establishment of the sown mixes. In practice, we would expect the rain garden bottoms to be able to evaporate and infiltrate the ponded rainwater after rainfall events, while the marginal areas are often the most visible spots and are also important for runoff interception and infiltration. Therefore, the species diversity and seedling density in the two saturation zones must be guaranteed to maintain the functional diversity. Adopting more appropriate species that suit their unique moisture regimes may be a possible solution to enhance the performance of sown plantings. Seedlings may be more sensitive to waterlogging and drought than the established plants (Chaturvedi et al., 1995; Middleton, 2002).

Therefore, transplanting of mature plants may strengthen the community's tolerance to the water excess in rain garden bottoms and the limited available moisture in marginal areas. Planting technique is thus recommended alongside the method of sowing in-situ to obtain a better planting establishment in rain garden bottom and margin. Furthermore, increasing sowing rate is recommended in rain garden bottom and margin, as the higher sowing rate could increase the number of seedlings of the sown species to cover the possible mortality of desired species (Hitchmough et al., 2004), which may also increase the species richness in sown community in the two saturation zones.

We are not surprised to discover that most sown species had statistically equivalent germination in margin and bottom, as the experimental rain gardens never encountered truly anaerobism or severe drought throughout the whole study. We thus assume that the abundances and evenness of the appropriate sown species tolerating a wide range of soil moisture conditions would not show significant differences between the margin and bottom in properly engineered rain gardens under the typical UK weather and moderate temperate climates. Nevertheless, the noticeable preferences of moisture in *Deschampsia cespitosa* and the considerable seedling reduction in *Lychnis flos-cuculi* in rain garden bottoms compared to margins suggest that the stated balance could be hardly sustained in humid regions with possible constantly waterlogged rain garden bottoms or in arid regions with little available soil moisture through time. We thus suggest that the seed mix design should respond well to climatic variation. For instance, seed mixes for humid regions may adopt a greater proportion of wetland species to be spread in rain garden bottom, whereas the marginal areas in arid regions may require a greater proportion of species tolerate of severe and extended drought.

It is difficult to draw valid conclusion for establishment of the very rare unsown ruderals including *Galium aparine*, *Myosotis arvensis*, *Papaver rhoeas*, *Rumex obtusifolius*, *Sonchus oleracea* and *Trifolium repens*, which had presence in less than 5 out of 45 quadrats across the experiment. As stated previously, these species were also found randomly distributed across the experimental trials. A possible explanation for the unexpected germination and random distribution of these rare weeds in the experimental trials is that they were germinated from the wind-born seeds acquired by the contaminated compost mulching. We thus emphasise again that the adaptation of sterile mulching must be ensured for the future application.

The in-situ sown seed mixture developed its own characteristic group of species respectively on the ‘dry-wet’ rain garden saturation zones (i.e. margin, slope and bottom). We assume the distribution and abundance of the candidate species in typical rain garden profile is similar to that of wetland, in which the lower limits of their biomass and density in the community are determined by their sensitivity to soil moisture and the upper limits are determined by the competitive ability of co-occurring species (Blom et al., 1994). Therefore, we consider the identification of specific species or functional groups of species that may be valuable for urban rain gardens depends largely on not only their resistance to the practical rain garden moisture profile, but also their competitive dynamics within a community. In this study, investigation clearly shows that species having same preferences on specific moisture regime and weeding treatment regime tended to happily co-exist together. Similarly, Dunnett (2004) suggest that matching species with the same ecological strategies and with the similar corresponding site conditions is vital to create ecological compatibility. RDA triplot shows an evident weed competition against the desirable sown species. However, as suggested previously, the sown plantings had significantly greater abundances over the ruderal competitors. RDA triplot also shows that all sown forb species excluding *Valeriana officinalis* with no emergence across the experiment could co-exist peacefully in experimental communities, which indicates that the sown seed mix was successfully combined. It may also ensure the future development of the functional group and fundamentally maintain its ecosystem services over time.

However, it is little surprising that, compared to the three sown dominants including *Achillea millefolium*, *Deschampsia cespitosa* and *Leucanthemum vulgare*, the other sown species showed either significantly less number of seedlings per quadrat, or had presences in rather limited number of quadrats. In fact, such outcome is unexpected. Ideally, the adopted seed mix design is expected to establish a sown community with a balanced species composition that most sown species should have similar presences and evenness across the experiment to maximise the species diversity and structural diversity. We assume the vigorous dominants would tend to outcompete the other non-robust co-occurring species (e.g. *Filipendula ulmaria*, *Galium mollugo*, *Lotus pedunculatus* and *Prunella vulgaris*, etc.) and thus decrease the community diversity over time. We recognise the risk of interspecies competition to the community diversity, and therefore suggest to reduce the proportion of the robust and vigorous species in the seed mix and also to increase the amount of seeds in those weak competitors. It is also worth noting that the present investigation was based on the species presence/absence data in only one growing season, while long-term establishment and stability have not been determined. It is still possible that some of the recognised “complementary” species might be able to show more emergences on-site in the years to come. In this way, the species composition and dominant keystone species may be changed to alter the vegetation pattern, landscape impact and ecosystem services of the sown community. A future evaluation for the responses of the sown mixes to year effects would be helpful, and is therefore strongly recommended.

5.5 Conclusion

Currently, increasingly limited management inputs are available for urban landscape, while the conventional planting schemes may often require high installation costs and considerable maintenance inputs. In this study, the adopted seed mixes showed relatively good performances in establishment and visual amenity. The seeding rain gardens are considered cost-efficient and environmental sound, and may thus have great potential to be publically acceptable and are strongly recommended for the future application. However, we suggest the future application to adopt less proportion of vigorous species in seed mixes to avoid the potential severe interspecies competition between the sown species. We assume the keystone structure of the sown community is capable of reducing runoff generation. However, the runoff retention and detention in the experimental rain garden models were not formally assessed in this study. Runoff quantity and quality control in practical rain gardens adopted sown taxonomically diverse planting would be a future research direction to offer valuable insights.

Application of mulches had effectively increased the species richness and density of sown species in the experimental units, and is therefore strongly recommended in the future implementation. However, it is worth noting that the results did not show much improvement of the emergence of sown forb-rich mixes in practical rain garden conditions due to mulch depth (i.e. 20 mm and 100 mm in the present study), but only to the presence/absence of mulch. Greater mulch depth is thus unnecessary for future application for minimising the installation cost and also to reduce the risks of undesirable interspecies competition and potential eutrophication in receiving watercourses connected to the rain gardens.

No valid conclusion could be drawn on the effectiveness of felt mats on the sown plantings in practical rain garden conditions because of the potential contamination of the compost. We notice that the effect of the quality of compost for mulching to sown vegetation and urban plantings is often underestimated in practice, that weedy mulching are often applied on top of planting beds and may thus result in a high abundance of weeds to outcompete the desired species. Therefore, future study and application are strongly suggested to adopt appropriate sterile compost mulching.

Although most of the adopted sown species in this study were positively responsive to the experimental rain garden settings, we still notice the evident different planting diversity throughout the ‘dry-wet’ saturation zones due to the habitat heterogeneity caused by the moisture gradient in practical rain garden soil profile. We consider the rain garden slope would be the most important spot to increase the planting diversity and to strengthen basic ecosystem services. Therefore, mixing more appropriate species in the seed mixes tended to be used in slope is strongly recommended. Species richness and seedling density in rain garden margin and bottom need to be strengthened. Alternative techniques such as combining planting with sowing and increasing sowing rates may be considered to benefit the initial establishment of sown species. We also strongly recommend the seed mix design for rain garden margin and bottom should respond to climatic variation.

Nevertheless, due to the availability of the essential instruments, the on-site precipitation and soil moisture data throughout the ‘margin-slope-bottom’ rain garden profile were not obtained during the whole growing season of the sown communities.

The recorded soil moisture data outside of the experimental period still makes sense to describe the differences in moisture distribution throughout the ‘margin-slope-bottom’ rain garden profile. However, it is highly recommended to obtain the soil moisture data during the growing season of the sown vegetation in future studies, in order to find out the exact dynamic spatiotemporal distribution of soil moisture in the rain garden margin, slope and bottom.

Due to the time scope of this study and its focus on the response to planting establishment techniques rather than the longer-term stability, the time effects on the establishment of proposed sown forb-rich communities in rain gardens were not studied. Additional research is thus required to understand the yearly effects on the sown rain garden seed mixes and the competitive interaction between species in different saturation zones. The visual characteristics of the developing sown forb-rich plantings for rain gardens is also interesting and is suggested to be studied in a multi-year study.

Chapter 6: Conclusion

In this thesis, Chapter 3 tested the success of plant establishment for a range of potential native and non-native perennials with different traits (i.e. forbs and grasses) under the effects of typical cyclic flooding and potential long-term drought in rain gardens. Chapter 4 studied the runoff retention and detention performance of the taxonomically diverse planting mix suggested from Chapter 3, and compared their effectiveness in rain garden hydrology with conventional mown grasses and bare soils. Chapter 5 studied seeding rain garden as an alternative installation and the effectiveness of two simple weed control methods (the use of felt mat and mulching with different depths) on the establishment of forb-rich plantings in rain gardens. The interaction between the establishment of sown mixes and the typical moisture distribution throughout the depression structures in practical rain gardens was also studied in Chapter 5. These topics are often underestimated in current rain garden implementations, which may reflect the lack of information and support from previous studies, and thus challenge the professions when making informed planting decisions.

This PhD study therefore provides a solid link between the planting and engineering performance of rain gardens, and has meaningful implications for future application. The two studies introduced in Chapters 3 and 4 provide the means to accurately determine preferred plant species for typical rain garden moisture conditions and predict vegetation types for improving the hydrologic performance in rain gardens. The results of Chapter 5 provide meaningful data to inform low-impact weed control methods for the establishment of sown plantings in rain gardens, as well as to predict the ideal plant choices and implementation means for vegetation according to the moisture conditions in practical rain gardens.

6.1 Summary of key outcomes and implications

6.1.1 Chapter 3

In Chapter 3, the survival, physiological growth (i.e. shoot/root dry weight, plant height and spread) and health (i.e. plant stress determined from leaf chlorophyll fluorescence) of the potted plants of 11 flowering perennial species (European native *Caltha palustris*, *Iris sibirica* and *Thalictrum aquilegifolium*, American native *Amsonia tabernaemontana var. salicifolia*, *Gaura lindheimeri*, *Rudbeckia fulgida var. deamii* and *Veronicastrum virginicum*, and Asian native *Astilbe* 'Purple Lance', *Filipendula purpurea*, *Hemerocallis* 'Golden Chimes' and *Sanguisorba tenuifolia* ‘Purpurea’) and four grass species (European native *Deschampsia flexuosa* and *Molinia caerulea*, and Asian native *Calamagrostis brachytricha* and *Miscanthus sinensis*) were measured under the effects of simulated cyclic flooding (i.e. repeated cycle of inundation to substrate level for 0, 1 and 4 days and 7-days draining) and extended drought in a ventilated greenhouse from 28 May to 28 June 2013.

This analysis concluded that the responses of perennial species to typical cyclic flooding in rain gardens are fundamentally determined by their moisture sensitivities and original habitats, whereas species performance appears to be not responsive to their geographical origins. We therefore suggest that the species recommended by most rain garden design manuals as those which could withstand continuously or periodic inundation may have the best tolerance and establishment in rain garden cyclic flooding conditions. We also recommend practitioners to adopt species from damp habitats such as swamps and water boundaries at the bottom and on the slope of rain garden depressions. Although 100% survival was found in all species in the cyclic flooding treatments, perennial species that were assumed to withstand only infrequent inundation showed variable tolerance levels to cyclic flooding. For example, *Hemerocallis* 'Golden Chimes' and *Iris sibirica* showed the best cyclic tolerance amongst all 15 candidate species, whereas *Amsonia tabernaemontana var. salicifolia* is only tolerant in short-term cyclic flooding (1 day). *Gaura lindheimeri* was the only species that was assumed to be intolerant of inundation. However, the results show that it was well adapted to both 1-day and 4-day cyclic flooding. Results show that all candidate species could happily establish in the 1-day cyclic flooding, whereas the longer-term (4d) cyclic flooding condition could significantly hamper plant health in a few adopted species. We thus also suggest a proper soil engineering to improve the rain garden infiltration and discharge to increase the plant diversity.

The physical growth of most species, excluding *Thalictrum aquilegifolium,* was significantly stunted by the effects of extended drought. Mortality was found in most species during the extended drought. However, this was considered to be acceptable because of the elevated temperature in the greenhouse throughout the whole study. In this study, perennial species assumed to be intolerant of inundation or able to withstand only infrequent inundation showed the best drought tolerance amongst the other groups. Therefore, these species are suggested to be planted on the rain garden margins or to be mixed with other species to improve the drought tolerance of the plant community. Perennials that grow naturally in habitats with continuous inundation and most species that are assumed to withstand periodic or seasonal inundation were determined to have relatively poor drought tolerance, and are therefore not recommended to be planted on the margins of rain gardens with droughty soils or in arid regions with rather low annual rainfall inputs. Similarly, plant species geographical origins were determined to have no significant impact on tolerance of the potential extended drought in rain gardens. Therefore, plant selection does not have to be restricted by ‘native only’ rules.

Chapter 3 provides data which could be used to inform the future planting for rain gardens. We expect the conclusions may allow the designer to choose appropriate plant species and to determine their distributions throughout the ‘wet-dry’ moisture gradient from the rain garden bottom to the margin. Successful plant selection will sustain the health of rain garden vegetation, as well as reduce maintenance and labour input.

In addition, previous studies often tend to use only survival and physical growth of plants to evaluate their performance in sustainable stormwater management facilities, which were unable to detect the stress a plant suffered in responding to cyclic flooding or drought. Determining plant stress could provide additional insight for effectively reflecting the potential damage in plant tissues due to water stress in advance of the visual damage, and therefore is valuable for predicting the long-term tolerance of plants to waterlogging and drought stress in typical rain garden conditions. However, traditional techniques of plant stress detection such as the measurements of the fresh weight/dry weight ratio in leaves, starch content, micro and macro nutrients, leaf and root electrolyte leakage and respiratory rate, etc. (Percival & Dixon, 1997; Figueiredo, 1985) are usually costly and destructive, as well as requiring a relatively long period of time to obtain data. In this study, leaf chlorophyll fluorescence was easily obtained by instrumentation and is a very effective tool to reflect plant stress caused by cyclic flooding and drought. It is therefore strongly recommended to be used in future research and applications for monitoring the performance of candidate species in rain gardens. Furthermore, the evaluation of the success of each plant species under cyclic flooding and drought treatments in this study coupled with the results of survival, physical growth and water-stress, which has led to holistic evidence of plant success in rain gardens, and are thus recommended to be adopted in future research.

6.1.2 Chapter 4

In chapter 4, a comparative study on stormwater retention and detention was carried out between bare soil (i.e. the non-vegetated control group), conventional mown grasses and the more diverse vegetation to assist designers in selecting vegetation types for rain gardens. In this study, the mown grass was created by sowing a commercial mix in-situ, which was a mixture of six lowland species including *Agrostis capillaris, Alopecurus pratensis, Anthoxanthum odoratum, Cynosurus cristatus, Deschampsia cespitosa, Festuca rubra*. The taxonomically diverse forb-rich plantings were created by transplanting pot plants of eight perennial forbs (*Amsonia tabernaemontana var. salicifolia, Astilbe* 'Purple Lance', *Filipendula purpurea, Hemerocallis* 'Golden Chimes', *Iris sibirica, Rudbeckia fulgida var. deamii,* and *Sanguisorba tenuifolia* ‘Purpurea’) and two grasses (*Calamagrostis brachytricha* and *Molinia caerulea*) that were determined to be suitable for typical rain garden conditions from the previous study in Chapter 3.

In this study, 15 rain garden models planted with different vegetation types were created. Ventilated rain shelters were established over the top of the experimental units to prevent the influences of natural precipitation but without greatly altering the evapotranspiration during the experiments. Artificial rainfall with known amounts and a controlled inflow rate was applied to the 15 rain garden models to process the runoff retention and detention tests. In this study, the assumed ratio of impermeable drainage area to rain garden was 5:1. 1 h rainfall of 21.94 mm depth was given to each rain garden model for the retention tests, which took place three times under different antecedent dry weather periods (ADWP, 2 days, 5 days and 7 days) from 8th September to September 23rd, 2013. 1 h rainfall of 72.6 mm was added into pre-wetted systems at 1.21 mm/min on three different days (September 25, 27 and 30, 2013) to allow monitoring of rain garden detention.

Retention analyses indicated that the taxonomically diverse forb-rich plantings with high species richness and structural diversity have significantly greater runoff reduction compared with the bare soils and mown grasses. Correlations between the amount of runoff reduction in rain gardens and the length of ADWP were found, where longer ADWP promoted the runoff retention capacity. In this study, the control group with bare soils retained 66.51%, 71.60% and 75.44% of the simulated rainfall under the 2-day, 5-day and 7-day ADWP, respectively. Mown grasses retained 69.70%, 70.70% and 71.85% of the artificial rainfall under the 2-day, 5-day and 7-day ADWP, respectively. Systems planted by the mixed forb-rich perennials retained 74.59%, 75.93% and 76.73% of the 21.94 mm artificial rainfall under the 2-day, 5-day and 7-day ADWP, respectively. We consider the experimental models had no infiltration, as the systems were sealed by impervious liners so that most runoff would escape from the outlet ports. Therefore, we guess the systems restore their retention capacity depending mainly on evapotranspiration during the ADWPs and absorption provided by vegetation and soils. It is worth noting that runoff retention in mown grasses was significantly less than the mixed forb-rich plantings in all circumstances and was outcompeted by bare soils after a longer duration of ADWP (5 days and 7 days). This indicates that the traditional mown grasses could only contribute rather limited retention capacity to sustainable stormwater management components. All the considerations led to the recommendation that the conventional rain garden vegetation that lacks plant diversity, such as mown grasses, should be replaced by taxonomically diverse plantings to provide better stormwater quantitative control and prevent urban catchments and public drainage systems from flash flooding.

In the detention tests, the longest t_{50} delay (i.e. the lag time between the medium value on the cumulative rainfall profile and the same absolute depth on the cumulative runoff profile) averaging 38.18 min and the lowest peak runoff flow rate at 0.85 mm/min (i.e. 29.75% percentage Peak Attenuation) were found in the mixed forb-rich plantings. Bare soils had the shortest t_{50} delay (24.74 min) and discharged runoff with the greatest outflow rate at 1.20 mm/min (i.e. 0.83% percentage Peak Attenuation) amongst the three different treatments. Mown grasses contributed to a mean t_{50} delay of 29.02 min and discharged runoff with a peak flow rate at 1.12 mm/min (i.e. 7.44% percentage Peak Attenuation). The greater detention performance in taxonomically diverse plantings is assumed to be improved by the greater interception for rainfall resulting from their greater canopy % cover and diversity of canopy structure, as well as the greater rooting depth and root expansion to reverse soil compaction to create more soil micropores to temporarily capture more water, so that the rainwater can move more slowly through the systems. The conclusions again indicate the advantages of the taxonomically diverse plantings in urban stormwater management over the conventional vegetation.

6.1.3 Chapter 5

In this study, a swale was divided into 15 sections to enable vegetation trials in rain garden conditions, in which an in-situ sown seed mix consisting of 15 British native perennial species (*Achillea millefolium, Cardamine pratensis, Centaurea nigra, Deschampsia cespitosa, Filipendula ulmaria, Galium mollugo, Leucanthemum vulgare, Lotus pedunculatus, Lychnis flos-cuculi, Lythrum salicaria, Prunella vulgaris, Rumex acetosa, Succisa pratensis, Thalictrum aquilegifolium,* and *Valeriana officinalis*) were allowed to grow naturally without any artificial irrigation from December 2013 to September 2014. Three pre-treatments: the use of felt mat (with or without) and mulching of green waste compost (two depths: 20 mm and 100 mm), and a non-treated control group were involved before sowing seeds in-situ as simple weed control means for the sown plantings. On 19 September 2014, the emergence of the desirable sown species and unsown ruderal species, as well as the % cover of vegetation per 0.5 $\times$ 1.0 m quadrat including all the sown and unsown ruderal species was obtained on-site to evaluate the effectiveness of the two weed control means.

From 16 September to 18 December 2014, the on-site soil moisture content in the margin, slope and bottom of a randomly picked experimental unit was instrumented to determine the interaction between the sown seed mixes and the ‘wet-dry’ saturation zones throughout the rain garden depression.

Results suggested positive establishment of sown mixes in practical rain garden conditions. We consider the seeding rain garden as a satisfying installation, which is therefore strongly recommended for the future application. The use of felt mat and mulching contributed to greater % cover of vegetation, while the mulching was determined to have the dominant effects on the establishment of the sown vegetation. Experimental units that adopted the weed control treatments had significantly greater total number of sown species and fewer emergences of unsown ruderal species per quadrat compared to the non-treated experimental units, which indicates the success of weed control from the application of the two simple weed control means. Analyses suggest that the application of mulching had significant effect on increasing the amount of seedlings in some of the desirable sown species including the dominated species such as *Achillea millefolium* and *Leucanthemum vulgare*, as well as significantly reduced the emergence of the dominant unsown ruderal *Chamerion angustifolium*. The result matches the hypothesis that the application of mulch in rain gardens allows a head start for the establishment of the sown plantings to build competitive advantages to outcompete the unsown ruderals. Therefore, the adoption of mulching is highly recommended for the future implementation of sown plantings in rain gardens. However, greater mulch depth was determined to have no effect on increasing the species richness and density of desired sown species for the sown forb-rich mixes, and is thus unnecessary in future application. No significant effect of felt mat was determined to improve the emergence of individual sown species, which is to be expected, as it was assumed to act as a barrier on the top of soil to suppress the emergence of unexpected weedy species. However, it is surprising that the stated results indicate that the use of felt mats had no significant effect on reducing the emergence of all the unsown ruderal species due to the contamination of potential weedy compost mulching.

Generally, there is evidence of a ‘wet-dry’ moisture gradient in the rain garden saturation zones, in which the bottom was determined to constantly have the greatest soil moisture content amongst all three saturation zones, the slope had a moderate moisture between bottom and margin, while the margin was the driest zone in the rain garden. Moisture distribution in saturation zones was determined to have a significant effect on the % cover of sown vegetation and the emergence of individual species. Slopes with moderate moisture had significantly greater % cover, total number of different species per quadrat and total number of seedlings per quadrat than that from the margins and bottoms. Individual species data shows a conclusion that appears to match the community performance. We consider the limited moisture availability in rain garden margins and the repeated multi-day inundations in rain garden bottoms were the vital factors to decrease the emergence of individual species.

We confirm that species with similar habitat requirements tended to happily co-exist together in the sown communities. In this study, establishments of all the sown forb species appear to positively relate to each other, which again suggests the success of the sown mixes. However, the sown community seems to be largely dominated by rather few vigorous species. Although we assume that some of the adopted species would show more emergences in the years to come, it is still possible that the competitive risk derived from the robust competitors may hamper the community dynamics.

6.2 Summary of novel contributions of the present study

In chapter 3, we thoroughly compared the between-species performances and suitability of a range of perennials in expected rain garden moisture profile. The candidate species were claimed to hail from different geographical origins. As previously noted, there is indeed limited selection of plant species suggested for urban rain gardens, which thus severely limit the species diversity, functional diversity and visual amenity for the rain garden type of vegetation. Plant selection is further restricted by the “native-only” adoption rule when ecologists are involved. In this present study, we confirm that the geographical origins of species have no significant impact on their tolerance to cyclic flooding and potential prolonged drought in rain gardens. This would thus allow researchers and landscape practitioners to test a wider range of potential species from the reference communities found in different geographic regions of the world.

There is a long recognised gap between the scientific researches and the real-world practices on the species selection for rain gardens that the scientific reports often tend to be incapable of guiding practices, whereas inappropriate species are often adopted in practices leading to increase of maintenance costs or even failure in vegetation establishment. For instance, perennial forbs are much preferred by landscape practitioners for their visual appeals and functional diversity. However, many forb species were adopted following recommendations that are based on speculation rather than scientifically tested evidences. Nevertheless, it is noticeable that the previous scientific studies of plant suitability in rain gardens only reported little range of species, in which the focus was mainly on shrubs and grasses while species diversity often tended to be ignored. In Chapter 3, a large proportion of the studied perennials were forb species, which also adopted sufficient species diversity to provide us confident data to confirm their suitability. In this study, we confirmed the existing expectations with respect to which ecotype of plants would be suitable for typical rain garden moisture profile. We recommend adopting the perennial species that are suggested to suit the periodic and infrequent inundations in current rain garden manuals for the periodically inundated rain gardens, while the species tolerating of infrequent inundations may maintain the functional diversity of the community during the dry spells.

Previous studies tended to use physiological growths of plants as the only indicators to determine the species suitability in specific rain garden conditions. In Chapter 3, we also detect the plant stress via chlorophyll fluorescence by hand-held instrument. The time series measurements of chlorophyll fluorescence provide additional insight for reflecting the potential damage in plant tissues due to water stress in advance of the visual deterioration. This approach is therefore considered valuable for predicting the long-term tolerance of plants to waterlogging and drought stress in typical rain garden conditions. We also coupled plant survivals and physical growths with the measures of water-induced stress in candidate species to provide holistic evidence of plant success in typical rain garden conditions. The comprehensive analysis developed robust quantitative criteria for plant selection and the determination of plant distribution throughout the ‘margin-slope-bottom’ rain garden profile.

Hydrologic performance of rain garden systems is a very active research topic. Most studies focus on the contribution of the integrated rain garden system, whereas rather limited quantitative studies successfully acknowledged the contribution of vegetation. Remaining unclear effects of vegetation types, and the highly variable observations in the retention and detention of stormwater runoff between the taxonomically diverse communities and conventional mown grasses also leave great research gaps. The stated research gaps can greatly challenge the public engagement due to doubt of the functionality of the taxonomically diverse plantings, and challenge professions to make informed planting decision. In Chapter 4, we thoroughly carried out a systematical comparison between the hydrologic performance of the species-rich perennial mixes and the conventional mown grass swards, and also separate out the retention and detention by allowing different antecedent dry weather periods (ADWP) and pre-saturation to experimental systems. This study allows us to collect meaningful data without interference due to the frequently ignored effects of ADWP and variable experimental conditions found in other previous studies. We confirm that the forb-rich perennial mixes may greatly improve the runoff retention and detention over the mown grasses due to their larger canopy and root growths and the much more complex structural diversity derived from the greater species richness. We are thus able to develop a meaningful planting framework to guide the future design of urban rain garden vegetation.

As previously stated, we notice that some of the leading figures in urban rain garden application started to adopt sown communities as an alternative installation method. However, most scientific researches tended to report the effectiveness of sown mixes with rather limited species-richness, while the focus was mainly on sown grasses which have been long-criticised by their poor visual appeals and restricted functional diversity. In Chapter 5, we are glad to have the opportunity to carry out the first-time experimental replicated trials of sown vegetation in practical rain gardens under the UK weather. The sown mixes provided satisfying emergences and establishment, which also provided pleasant flowering display, as well as were assumed to have satisfying functional diversity and to be capable of reduce on-site runoff generation. We thus consider the seeding rain garden as an effective, environment-sound and cost-efficient installation for the future application.

Little studies had mentioned the effects of maintenance regimes on the suppression of weediness in rain garden sown plantings. Felt mats and compost mulching involved in this study are two less interventionist weed control approaches compared to other applications such as applying herbicides, seasonal cutting or removal of productive topsoil. Although evident weediness was observed across the experiment, the experimental outcomes clearly suggest the effectiveness of the adopted weeding treatments. It is surprising that felt mats did not show significant effects on suppressing the indigenous ruderal species. We consider the mulched composts might acquire a significant load of wind-born seeds to contaminate the experimental units, and thus suggest the application of sterile mulching in future practice.

We found the previous studies in respect of rain gardens or similar stormwater management facilities (e.g. bioswales) tended to explain the planting success or failure by the tolerance of selected species, whereas no previous studies had clearly reflect the effects of the dynamics of moisture distribution in rain garden soil profile on the establishment of in-situ sown seed mixes. In Chapter 5, statistics confirmed that the ‘dry-wet’ moisture gradient throughout the ‘margin-slope-bottom’ rain garden profile showed dominant significance in altering the vegetation patterns and the distribution of specific species over weeding treatments. Slope with the moderate available moisture content between the damp bottom and the dry marginal area contributed to the greatest species richness and seedling density of the sown species, and is therefore considered the perfect spot to increase species diversity for the sown mixes. Repeated inundations in rain garden bottom and the potential water scarcity during dry spell are assumed to be the dominant constraints to restrict the performances of sown mixes. We therefore recommend transplanting mature plants and adopt tolerant species to strengthen the resistance of the sown communities to the potential water stress in rain garden bottom and margin.

6.3 Limitations

6.3.1 Chapter 3

One limitation of the experimental framework is that there was no device to carefully control the temperature and air humidity in the ventilated greenhouse. The elevated temperature and variable air humidity were acknowledged to be two potential factors to alter the plant stresses in a few species. During the 32 days of cyclic flooding tests, all candidate plants tended to grow very fast to their maximum due to the elevated temperature in the greenhouse. The results are therefore valuable for predicting their further adaptations to cyclic flooding treatments.

It should be noted that although the physical growth of *Thalictrum aquilegifolium* was not significantly affected by cyclic flooding treatments throughout the whole duration of this study, continuously decreasing chlorophyll fluorescence and rather poor stress tolerance during flooding treatments were detected in this species. Thus, potential damage in this species might become visible if more flooding cycles were applied.

In cyclic treatments, the 4-day draining would allow the experimental soil to return to moderately moist conditions before the next flooding. It should be noted that these conclusions are based on controlled experiments. In practice, rain gardens may also experience weather shocks such as a rapid switch between extended drought causing extremely low moisture content in soil and instant saturation resulting from large storm events or flooding.

We consider the cyclic water baths and withheld of irrigation for containerised plants succeeded in simulating the expected cyclic flooding and potential extended drought in urban rain gardens. However, we notice that the limitations in container-grown plants, such as the restricted root extension and the limited soil water content for each specimen due to the little available soil volume imposed by used pots, may somewhat alter the species performances compared to that of the real-world practices. We thus suggest a comparative study to monitor the species development in practical rain gardens for the future study.

Moreover, due to the loss of substrates from pots in deeper inundation over substrate level, the cyclic flooding experiment was unable to determine the effects of periodically deeper inundations (e.g. flood to leaf level in plants) on plant growth and health. Due to time constraints and funding issues, this study is also limited by a lack of diversity of plant species, in which the success or failure of only a small sample of plant species (i.e. 11 flowering forbs and 4 ornamental grasses) was studied.

6.3.2 Chapter 4

It is very obvious that the phenology of plant canopy could greatly change in hydrologic dynamics in rain gardens, however, this study only monitored the systems' runoff retention and detention in autumn instead of multiple growing seasons and the yearly changes. Canopy interception is also assumed to greatly alter the hydrologic dynamic in rain garden vegetation (Dunnett & Clayden, 2007), although this was not explicitly detected in this study. In this study, the aims were to study the differences in runoff retention and detention from different vegetation types under the effects of extreme storm events in a short duration (e.g. the 1 in 10 years and excess of 1 in 50 year events for the FEH design storms). However, it would be worth finding out the systems' performances with a range of stormwater input sizes, which would increase the utility and accuracy of models predicting the rain garden hydrology.

Moreover, although the aim of the present study is to quantify the contributions of different vegetation types to rain garden hydrologic performance, it would also worth looking at the alteration in the retention and detention offered by the choice of single species. The original intention to establish the mixed forb-rich plantings by sowing seeds in-situ to gain naturalistic appeal and a random planting pattern, as well as complicated aboveground traits. However, it is likely that the structurally complex community may require several years to reach the desired establishment from the seeding method than transplanting, which therefore has not been done due to the time constraint of the study.

6.3.3 Chapter 5

In this study, seed mixes were sown in December 2013, in which minimal wind-born seeds could influence the sown vegetation. However, the attempted testing of the felt mats was not successful. The results demonstrate that the use of felt mats have no significant effects on supressing the unsown ruderal species. However, this failure may not indicate that the felt mats have no effectiveness in supressing weeds from the existing soil seed bank. Two experimental observations that the greater mulch depth significantly promoted the % cover of the vegetation comprising of sown species and ruderal species and there is no significant effects of greater mulch depth on increasing the species richness and density of the sown communities led to the suspicion that the mulch was not weed-free. Although the used green waste compost was determined to be weed-free on 11 October 2013, the main concern is that the composts were exposed in the outdoor environment during October and November 2013, and thus were assumed to acquire a great number of wind-born seeds. Most of the ruderal species found on-site were assumed to grow from the potentially weedy composts, and is therefore assumed to fundamentally alter the effectiveness of felt mats. No valid conclusion of the effectiveness of the felt mats on sown plantings in rain garden conditions could be drawn from the present study.

The on-site soil moisture data obtained from 16 September to 18 December 2014 is valuable to evaluate the differences in moisture distribution throughout the 'margin-slope-bottom' saturation zones in the experimental rain gardens. However, another limitation is that, due to the availability of the essential instruments, the on-site soil moisture data was not obtained during the growth seasons before September 2014, when plant observations were taking place.

The seed mixes in this study included a number of suggested British native species that were available via local nurseries. There are many potential non-native species, which are assumed to be suitable for the sown vegetation in rain gardens. However, the potential exotic species were not used due to either the higher price of seed or their limited availability in the British market. Moreover, the time effects on the establishment of the sown forb-rich plantings were not studied in the present study due to time constraints.

6.4 Future work

To further understand the interaction between plant growth/health and the cyclic flooding/potential drought in rain gardens, ideally future studies should be carried out in a properly controlled environment with stable indoor temperature and air humidity. It is worth providing more flooding cycles to find out whether plants would establish their long-term tolerance to cyclic flooding conditions or let the potential damage in plants become visible. It is also worth testing plants in repeated cycles of extremely low soil moisture and instant saturation, as well as irregular moisture to identify their tolerance to weather shocks. Ideally, depressions of typical rain gardens are very shallow, which are normally designed to dewater in a short period of time, so that deeper inundations to foliage level in plants would be rare. However, periodically deeper inundations are expected at times, especially for the plants grown at the depression bottom, it is therefore worth flooding the plants intended to be planted at the bottom of rain garden to their leaf level in cyclic flooding treatments in future study.

To better understand the fundamental mechanisms of retention and detention in rain gardens, it is suggested that a further study be carried out in the practical rain gardens introduced in chapter 5, in which natural precipitation and infiltration is allowed, as well as having the chance to obtain the hydrological performances of in-situ sown taxonomically diverse plantings. In future studies, the runoff reduction contributions derive from evapotranspiration and soil infiltration, as well as soil absorption should be properly instrumented. For example, direct measurements of soil moisture content changes during dry weather would allow losses through evapotranspiration to be estimated and compared with the observed retention depths. It is worth looking at the influences of plant phenology and yearly changes in rain garden retention and detention in practical conditions. It is also worth establishing more small-scale rain garden models planted with single plant species to find out how different species may alter the runoff retention and detention, and therefore provide more suggestions for the planting choices.

For future study and application of sown plantings in rain gardens, weed-free mulching should be provided to minimise the impact of wind-born seeds. The effectiveness of felt mats is therefore suggested to be re-tested with the application of sterile composts. In future studies, the on-site soil moisture in rain gardens should be obtained during the whole duration of growing season of the sown vegetation. It is also very valuable to determine the year effects on the dynamics of the sown forb-rich plantings.

The main focus of this PhD study was to look at the plant selection for proposed taxonomically diverse rain garden vegetation, their hydrologic performances, the potential low-impact weed control means for sown rain garden plantings and the interaction between the sown plantings and the moisture gradient in rain gardens. In future studies, it would be valuable to explore the visual characteristics of the developing sown vegetation in rain gardens and the feedback from the visiting public to make informed decisions in planting design for greater aesthetics. Ecology of the sown taxonomically diverse rain garden vegetation is another important aspect to look at, such as their contributions to the urban biodiversity.

The present studies provide the conclusions that are capable of predicting plant species and vegetation types for pre-construction rain gardens based on their moisture requirements and hydrologic performances. However, only limited plant species and plantings were studied in this PhD. It is worth noting that there are a great number of non-native species and potential reference communities across the world, which are assumed to be suitable in typical rain garden conditions and are admired for their aesthetic appeal and ecological benefits. Due to the time constraints and limitation in funding, no potential non-native species were used in the sown seed mixes. A possible research topic for future study is to look at the feasibility of those potential reference communities and the non-invasive exotic species in practical rain garden conditions under the British weather, and to explore their hydrologic performance, aesthetics and ecological benefits.

6.5 Implication towards future design practice

Urban rain gardens to date are often engineered without enough consideration of horticulture and ecology. A consideration of plant diversity as one of the most important factors governing the basic functions (e.g. stormwater management) and ecosystem services of rain gardens is missing in urban rain gardens. In this PhD, the first aspect confirmed existing expectations with respect to the appropriate plant selection criteria for typical rain garden conditions, as well as which plants would be best suited to the bottoms, slopes and margins of periodically-inundated rain gardens. Another important implication of the first study is that it indicates that there are many potential species with different traits and different geographical origins that could be adopted for the development and improvement of urban rain gardens. A much broader selection of plants could allow designers to develop a higher diversity of vegetation in rain gardens, which has greater structural diversity and evenness of the abundance of plant species compared to conventional urban vegetation such as mown grasses. Such taxonomically diverse planting was determined to significantly improve the runoff retention and detention of rain gardens in the second aspect of the work.

Therefore, not only further exploration of the potential species from the reference communities across the world should be carried out, but the future design practice of urban rain garden vegetation should also take a step forward to embrace diversity. Three major concerns derived from the present study may provide a simple design framework for the taxonomically diverse plantings in urban rain gardens:

(1) Planting design in urban rain gardens should achieve a sufficiently

complicated aboveground structure. It requires the use of plants that have different heights, various canopy properties and different foliage shapes, in order to form a complex multi-layer structure. It would not only strengthen the runoff retention and detention, but may also improve the aesthetic value (Fig. 6.1).

(A) (B)

Fig. 6.1. Taxonomically diverse planting established in the second study (B) improves the aesthetics in rain gardens compared to the conventional mown grasses (A). Photo was taken by the author in August, 2013.

(2) Rain garden vegetation may adopt as many suitable plant species as possible, in order to maintain the complexity of its structure. In the vegetation mixes, some species may have a relatively short growing season, so that a dramatic change of community structure may occur when their tissues have died. Increased species richness could allow other species to grow and occupy the spaces of dead species, so that the spatiotemporal dynamics and balance of plant community structure can be sustained.

(3) ‘Right plant, right place’. Establishing plants in planting positions

appropriate to their nature and ecological needs could minimise the input for the site preparation and maintenance that they would require (Hansen and Stahl, 1993). The present study indicates that the successful plant selection for urban rain gardens is based on a full recognition of the habitat requirement for each candidate species, and the variability of soil moisture conditions which could be assigned in the typical depression structure of rain gardens. In general, the distribution of plants in rain garden saturation zones are significantly affected by the frequency and duration of the cyclic inundation and the moisture gradient throughout the ‘margin-slope-bottom’ spatial profile. For instance, upland species are not recommended in basin bottom due to the potential multiple inundations. Obligate wetland plants are inappropriate for using on the marginal area and upland of rain gardens, or the sunny slope with sandy soils, as these spaces may dewater rapidly after rainfall and stay dry until the next precipitation. Although the sunny/shady conditions were not discussed in this PhD, the observations in the third study demonstrated that the amount of sun and shade on site could greatly alter the structural diversity and density of sown vegetation. Plants needing full sun are thus not recommended to be planted at a shady site adjacent to high canopy trees or hedges, or in the shade of a distant building.

(4) The adoption of suitable plant species is also important to the runoff retention and detention in rain gardens. This PhD mainly focuses on the determination

of hydrologic performances of different planting types, where the results suggest that more diverse planting tends to improve stormwater management. Although the contribution of specific single species on the hydrologic performances of rain gardens has not been particularly studied, it is clear that plant species with larger spreads and greater heights would contribute to a greater runoff retention (Nagase & Dunnett, 2012), and plants with deep roots and greater root expansion could greatly improve the infiltration of systems. Therefore, it is suggested that the designed taxonomically diverse planting should adopt as many such suitable species as possible.

As stated previously, the concept of the taxonomically diverse planting is inspired by the natural reference communities. A next step to evolve planting design in urban rain gardens is to seek instructions from nature. There are many naturally taxonomically diverse communities with great structural diversity and species richness in nature, which also provide great amenities. For example, the wild moorlands in the Peak District, Derbyshire, UK, predominated by the fully hardy *Calluna vulgaris* and *Juncus acutiflorus*, are rich in plant diversity and sustain the landscape impact and native biodiversity over time, and might also serve as a reference community for urban rain garden vegetation in the UK (Fig. 6.2). *Calluna vulgaris* may not tolerate being submerged by water and therefore would be planted on the margins of the rain garden, while *Juncus acutiflorus* would perform well at the bottom and side-slope of the rain garden. Therefore, a possible way to achieve the taxonomically diverse plantings with great structural diversity and species richness in urban rain gardens is to adopt the potential species and species composition from the natural reference communities.

Fig. 6.2. Naturalistic swale with spontaneous *Juncus acutiflorus* growing in lowland and *Calluna vulgaris* on the swale slope and upland, Peak District, Derbyshire, UK. Photo was taken by the author in August 2013.

Nature as a guide could also turn the rain garden planting away from excessive and costly maintenance, such as the intensive long-term removal, thinning and weeding to maintain the tidiness of the vegetation. For example, in natural plant communities, size, structure and plant spacing are closely related, self-thinning occurs at every stage of vegetation establishment that the same populations begin as many small plants but end up as few large plants (Beck, 2013). The self-thinning principle is thus important to the adoption of taxonomically diverse plantings in urban rain gardens, so that the desirable plant species, regardless of their size, should be planted at high densities necessary to immediately crowd out weeds, and allow intraspecific competition to maintain the balance between plant density and plant size for species evenness (Beck, 2013). MacArthur and Wilson (1967) suggested that the countervailing forces of extinction and immigration could result in an equilibrium level of species richness in two isolated habitats if the migration corridor was restored between them. Therefore, the future design of urban rain gardens may employ the principle that functional habitats such as bioswales could be provided as the migration corridors between rain garden plantings, so that the desired biological and hydrological connection between rain gardens may naturally achieve the immigration of plant species and thus sustain the species richness in this network over time.

Mutualism between the city users and the taxonomically diverse planting is the vital factor governing the incentives for the application of such vegetation in urban rain gardens. Taxonomically diverse planting in urban rain gardens can reveal the aesthetic value of biodiversity by making it visible and operative as an interactive landscape element, which provides the best educational opportunities for residents to learn from on-site experience. However, only the aesthetics and ecosystem services may not govern the long-term commitment and motivations of city users on the use of such vegetation in rain garden, where the cost-efficiency and simplicity of application are also important to its public acceptance. Therefore, the third study in this PhD looked at the establishment of the low-input in-situ sown plantings in practical rain gardens.

As noted previously, the abundant invasive weeds from seed bank recruitment in urban soils could outcompete the sown species and thus adversely alter the ecosystem services and landscape effect of the sown plantings. The third study showed that compost mulching significantly increased the species richness and density of the desired sown species in the sown communities and had significant effect on suppressing the dominant ruderal species. Compost mulching is not only less interventionist compared to many other weeding approaches such as the application of herbicides and the removal of topsoil, but could also improve the soil fertility for the cultivation of the desired species. However, in the third study, the effects of the two simple weed control means were suspected to be greatly impacted by the contamination of the potential weedy compost mulching, while no valid conclusion could be drawn on the effectiveness of felt mats on suppressing weeds. Moreover, in practice, weedy composts are often mulched on top of the planting beds of urban plantings and may thus result in weed abundance to outcompete the desired species. Therefore, the adoption of appropriate weed-free mulching must be ensured in the future practices of sown plantings in urban rain gardens.

The third study confirmed that the different distribution of soil moisture content throughout the ‘margin-slope-bottom’ rain garden profile had significant effects on the establishment of the sown seed mixes. Therefore, it is worth making different mixes to suit the different saturation zones in rain gardens and involve transplanting techniques for the establishment of healthy sown communities in practical rain garden conditions. The side slope is the best saturation zone in rain garden depression structures to adopt in-situ sown plantings. It is worth mixing more species in the seed mixes intended to be sown in the rain garden slope to improve the plant diversity and landscape effect of the plant community. The low availability of soil moisture in the rain garden margin resulted in less plant diversity and density of desired sown species in the rain garden marginal area and upland compared to the moist slope. It is therefore suggested that seed mixes for rain garden margin should comprise more drought-tolerant species for gaining better establishment. Transplanting mature plants with better drought tolerance could also be adopted with sowing seeds in-situ to ensure the community density in rain garden marginal areas. The potential multi-day inundation in the wet rain garden depression bottom is assumed to be the vital factor to greatly reduce the species richness and seedling density of sown communities. In future applications, seed mixes for the wet rain garden bottom and the humid regions with significant annual rainfall inputs are required to have more inundation tolerant species. Increasing the sowing rate would be helpful to increase the emergence of seedlings in desirable species. Mature plants often have better tolerance to inundation compared to seedlings, therefore, transplanting could also be considered to improve the community’s tolerance to the waterlogged conditions in rain garden bottoms.

The original intention of the designed seed mix in the third study was to develop a community in which most sown species would have similar presences and evenness across the experiment to maximise the species diversity and structural diversity. However, the sown communities were largely dominated by the dominants during the period of the study, while low emergence of the proposed coexisting species were found. The sown community was thus relatively uniform low diversity compared to the community established by transplanting in the second study (Fig. 6.3). The major concern is that the sown communities may develop to be more diverse in the coming years as the seed mixes employed perennial species of which some may need more growing seasons to be established from seeding, whereas the transplanted communities in the second study had a greater evenness of the abundance of all the species than in the sown plantings. It is also likely that germination of these species was either adversely altered by the periodically-inundated conditions in practical rain gardens, or eliminated by interplant competition as the vigorous species may easily establish their predominances in the sown community. Therefore, planting could be considered for increasing the density of the beneficial minor species in the design community, while the proportion of seeds for those vigorous competitors should be reduced.

(A)

(B)

Fig. 6.3. Community established by transplanting in the second study (A) showed a greater structural diversity and dynamic appeal compared to the sown plantings established in the third study (B). Photo was taken by the author in August, 2013 and June 2015, respectively.

In this PhD, both planting and seeding are confirmed to be able to establish desired taxonomically diverse planting in urban rain gardens. Taxonomically diverse planting composed of various beneficial perennial species was determined to have advanced aesthetic value and hydrologic performance in stormwater management, and is thus undoubtedly having the potential to gain popularity in city inhabitants. In practice, people may consider the year round performance of the landscape impact and runoff attenuation in rain gardens as an important criteria for the success of the taxonomically diverse planting. The use of only perennial species excluding some evergreen grasses may die out after the growing season, and thus can’t meet the requirement to sustain their hydrologic performance, ecosystem services and appeals throughout the year. Therefore, evergreen perennials, shrubs and small tress should also be considered in design practice.

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