The efficacy of using a natural soil additive for the establishment, survival and diversity of native prairie and spontaneously colonizing plant communities on unirrigated green roofs in a humid subtropical climate

By

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Green roofs are an emerging technology promoted primarily for stormwater management but little has been published about their potential for biodiversity performance. This is the first study to explore the potential for creating prairie-like, non-succulent, native plant communities on unirrigated extensive green roofs in the southeastern United States. Ten experimental green roof platforms were used to: 1) identify native species and methods of establishment appropriate for green roof applications in the southeastern United States; 2) examine the effects of introducing natural soil into a commercially available green roof soil media mixture on the survival and establishment of native prairie species; and 3) examine the composition of early successional green roof plant communities. Eleven planted species were successfully established and 46 colonizing species were identified. It was found that the addition of native prairie soil did not significantly affect survival, overall cover, or biodiversity in terms of species richness and evenness.

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CHAPTER I

INTRODUCTION

1.1 Background

The term "green roof" generally refers to a rooftop designed to grow vegetation by installing a layer of lightweight soil media on top of a root barrier, filter fabric, drainage layer and a waterproof membrane (Getter and Rowe, 2006). Many of the environmental benefits that green roofs provide such as stormwater management, energy conservation and mitigation of the urban heat island effect are well documented (Getter and Rowe, 2006), however little investigation has been conducted regarding the role of biodiversity in green roof performance (Orbendorfer et al., 2007). Intensive maintenance regimes that include irrigation, weeding and fertilization are frequently recommended for green roofs but arguments have been made that these practices undermine goals of biodiversity and the long-term sustainability of green roofs. Additionally, there has been doubt as to whether or not native plants can survive on green roofs without supplemental irrigation (Monterusso, Rowe and Rugh, 2005) and as of 2012, examples of native, non-succulent plants grown on unirrigated green roofs in the United States are absent from the literature (Butler, Butler and Orians, 2012). In the United States, green roof technology has become more common in northern, central and eastern regions and is now being implemented in the South as well (Dvorak and Volder, 2013). Even though the greatest environmental gains from green roofs may be achieved in subtropical climates characterized by high day-time temperatures and intense rain events (Simmons and Gardiner, 2007), little is known about plant performance on green roofs in the South (Dvorak and Volder, 2013). In order for green roof technology to be implemented throughout the wide range of geographic and climatic conditions present in the United States, regionally specific research to determine appropriate plant taxa for use on green roofs needs to be conducted (Monterusso et al., 2005). Furthermore, there is a great need for green roof biodiversity research in ecoregions where biodiversity is in decline (Dvorak and Volder, 2010).

1.2 The Mississippi Black Prairie

The research for this thesis was conducted at Mississippi State University, located adjacent to Starkville in Oktibbeha County, which is in the east-central portion of Mississippi and is located within the physiographic region of the Black Land Prairie (USGS, 2004), also known as the Black Prairie (Leidolf and McDaniel, 1998). This area forms a crescent shaped band that extends from southwestern Tennessee at its northern extent, southward through east central Mississippi, and then east-southeast through central Alabama approaching the Georgia border (Schotz and Barbour, 2009). The region as a whole lies within USDA plant hardiness zones 7b and 8a (USDA, 2012). The Black Prairie near Starkville is bordered on the west by the Flatwoods and on the east by the Fall Line Hills physiographic regions (USGS, 2004). In 2012, Starkville was changed from USDA plant hardiness zone 7b (5° to 10° F, -15° to -12.2° C) to zone 8a (10° to 15° F, -12.2° to -9.4° C) The characteristic soil of the Black Prairie largely consists of a calcareous loamy-clay (Lowe, 1921) of thicknesses ranging from several feet to a few inches overlaying a layer of Selma chalk derived from Cretaceous limestone (U.S. Bureau of Soils, 1907). The Black Prairie region of Mississippi has been in agricultural cultivation since the early 1800's (Lowe, 1921) and since that time has been shifted away from a predominately grassland landscape (Leidolf and McDaniel, 1998). The few remaining prairie relicts in the area are threatened by cultural disturbance (excessive erosion and invasion by exotic weeds) and lack of natural disturbance, such as fire, which has resulted in the encroachment of aggressive woody vegetation (Leidolf and McDaniel, 1998). The area is floristically diverse with Oktibbeha county containing more state listed species of special concern than any other county in Mississippi (Leidolf, McDaniel and Nuttle, 2002) Leidolf and McDaniel's (1998) floristic study of a 41.5 ac (16.8 ha) disturbed and eroded Black Prairie relict, bordered by Eastern Red Cedar dominated woods interspersed with chalk outcrops and patches of open prairie, revealed 152 species of vascular plants, 7 of which were listed as imperiled and 4 more proposed as imperiled (Leidolf and McDaniel, 1998).

1.3 Purpose of the Study

This study explores the potential for creating prairie-like communities of native, herbaceous plants on unirrigated, vegetated extensive green roofs in the southeastern United States. The goals of this study are to: 1) identify potential native species and methods of establishment appropriate for unirrigated green roof applications in the southeastern United States; 2) examine the effects of introducing native soil into a commercially available green roof growing media mixture on the survival and establishment of native prairie species; and to 3) examine the composition of the resultant plant communities.

CHAPTER II

LITERATURE REVIEW

2.1 Environmental Benefits of Green Roofs

Soil and vegetation play an integral role in the hydrologic cycle. In undeveloped settings, they intercept and hold precipitation and allow for infiltration and groundwater recharge, evapotranspiration, and absorption of solar radiation (Getter and Rowe, 2006, Jennings and Jarnagin, 2002). In contrast, impervious surfaces intervene in these processes and cause environmental problems due to both increased quantity and decreased quality of the resulting runoff (Orbendorfer et al., 2007; Carter and Jackson, 2007; Jennings and Jarnagin, 2002). Impervious surfaces, such as parking lots and rooftops, also absorb solar radiation and re-transmit heat which results in increased atmospheric temperatures; a phenomenon called the urban heat island effect (Orbendorfer et al., 2007). Urban development usually produces vast areas of impervious surfaces and the excessive stormwater runoff has traditionally been channelized and routed to nearby streams and rivers. This practice increases the probability of downstream flooding as channel capacities are frequently exceeded and in cities that use the same infrastructure for both wastewater and stormwater, results in combined sewer overflows (Getter and Rowe, 2006; Orbendorfer et al., 2007). In streams and rivers, elevated flow volumes degrade aquatic habitats by scouring stream banks and channels, and accelerating erosion (Walsh, et al., 2005).

Green roofs are man-made ecosystems that consist of vegetation growing in a composite soil medium over a waterproof membrane (Orbendorfer et al., 2007) and are considered best management practices (BMPs) for stormwater management (Carter and Jackson, 2007). Green roofs are divided into two types: **intensive green roofs**, which are associated with substrates deeper than 15.2 cm (6 in), and are often used to re-create park-like landscapes on rooftops, and **extensive green roofs**, which are associated with substrates of 15.2 cm (6 in) and shallower (Getter and Rowe, 2006). Extensive green roofs are typically much less expensive to construct and maintain (Gedge and Kadas, 2005; Kadas, 2006). Hereafter, "green roof" will refer to extensive vegetated green roofs. Green roofs provide ten main benefits: reduced stormwater runoff, delayed stormwater runoff, reduced energy consumption, reduced urban heat island effect, increased biodiversity and habitat, mitigation of air pollution, increased lifespan of roofing membranes, reduced noise pollution, increased aesthetic value (Getter and Rowe, 2006) and carbon sequestration (Getter, Rowe, Robertson, Gregg and Andresen, 2009). But perhaps the most widely realized green roof service is the reduction of stormwater runoff volume, achieved by capturing precipitation and allowing it to be slowly released back into the atmosphere (Getter and Rowe, 2006, Orbendorfer et al., 2007). During rain events in which a green roof becomes saturated, runoff still occurs but at a slower rate because rainwater is delayed as it passes through the media (Getter and Rowe, 2006; Orbendorfer et al., 2007). These functions allow conventional stormwater management infrastructure to process lower volumes of runoff over a longer period of time (Getter and Rowe, 2006), therefore providing opportunities to mitigate erosion, sedimentation, and stream bank scouring, and in some cases pollution from sewer overflows.

Green roofs can also provide opportunities to improve the quality of stormwater runoff. According to the Water Resources Group (Getter and Rowe, 2006), two thirds of all impervious surfaces serve as vehicular infrastructure in the forms of parking lots, driveways, roads and highways. These surfaces collect pollutants such as oil, heavy metals, and salts which, in urban areas, are transported to natural waterways during rain events (Getter and Rowe, 2006). Separate stormwater BMPs are needed to intercept these pollutants but green roofs can help make their implementation more feasible by decreasing the volume of water they have to accommodate.

Green roofs can also provide opportunities to mitigate urban heat island effects. The urban heat island effect is a microclimate that is created primarily by parking lots, streets, rooftops, sidewalks and other hard surfaces that absorb sunlight during the day and reflect it back into the atmosphere at night as heat (Solecki et al., 2005). Green roofs work to mitigate heat islands through shading and evapotranspiration (United States EPA, 2008). Shading is performed by plant and soil media and reduces the amount of solar energy that is typically absorbed and reflected back into the atmosphere from conventional rooftops (Simmons, Gardiner, Windhager and Tinsley, 2008). Plants also actively cool the surrounding air through evapotranspiration (Getter and Rowe, 2006); a process resulting in water evaporating from leaves at the expense of heat from the air (United States EPA, 2008). Shading leads to further benefits as added protection from solar exposure protects the actual roof membranes from daily expansions and contractions, thus lengthening their lifespan (Getter and Rowe, 2006, Orbendorfer et al., 2007). Green roof soil and plant media provide added insulation which results in less energy consumption from heating and cooling (Getter and Rowe, 2006; Orbendorfer et al. 2007, Carter and Fowler, 2008). These last two functions account for more indirect ecological benefits in the forms of reduced energy and construction material consumption (Getter and Rowe, 2006).

A less understood green roof function is the capacity to increase urban biodiversity and provide habitat for plants and animals (Brenneisen, 2003; Coffman and Davis, 2005; Dunnett, 2006; Getter and Rowe, 2006). Some green roofs have proven to be very beneficial habitats in urban areas where green space is rare (Gedge, 2003; Brenneisen, 2003; Coffman and Davis, 2005) and may often remain undisturbed as many are inaccessible to the public (Getter and Rowe, 2006).

2.2 Scaling Green Roof Services

Green roof services such as reduced degradation of streams, urban heat island effect mitigation, avoided air pollution and reduced stormwater infrastructure needs are realized when the technology is implemented on a city-wide scale (Nui, Clark, Zhou and Adriaens, 2010). Accurately estimating these benefits has only become possible with recent advances in spatial analysis software such as Arc GIS used in conjunction with currently accepted hydrological modeling (Thurston, Goddard, Szlag, and Lemberg, 2003). Such analysis allows researchers to inexpensively gather large amounts of detailed geographic data and effectively identify sources of non-point source pollution (Thurston et al., 2003).

The next two sections detail exemplary studies concerning large-scale roof-greening. Section 2.2.1 details a study by Nui et al. (2010) to estimate the economic benefits realized by a city-wide roof greening of Washington D.C. Section 2.2.2 details a study by Carter and Jackson (2007) which implemented spatial analysis software to develop a strategy to direct roof-greening efforts in an urban Georgia watershed.

2.2.1 Estimating Economic Benefits

An interdisciplinary group of researchers looked at how green roof services could be realized at the city scale using Washington D.C. as a model (Niu et al., 2010). The researchers examined and estimated the annual monetary savings created by various hypothetical roof greening scenarios from reduced costs of stormwater infrastructure, pumping, and credits given for BMP installation based upon Washington D.C.'s municipal stormwater fees. They also extrapolated energy savings to the city scale from decreased electricity and natural gas consumption and estimated the impact of green roofs on the uptake of nitrogen monoxide (NO) and nitrogen oxide (NO2) based on experimental models (Nui et al., 2010). Lastly, they incorporated the collective savings into an economic framework model in order to calculate a range of points at which the savings break even with the added cost of green roof installation as compared to conventional roofs.

Washington D.C. has implemented a stormwater management fee which more accurately charges for stormwater generation based on impervious surfaces (Nui et al., 2009). The fee shifts the cost burden of stormwater management toward large stormwater generators and away from other property classes such as residential. Included in the revision is a policy of granting a 35% or a 50% reduction in stormwater fees for installing a green roof. By aggregating these discounts at the city scale, Nui et al. estimated an annual savings range from \$0.22M to \$0.32M. In terms of stormwater reduction, they estimated annual operational savings of \$0.95M and the infrastructure size reduction savings over 40 years to be \$13M. Natural gas and electricity savings were based on studies that compared the consumption from buildings with and without green roofs. These reductions were aggregated to the city scale and resulted in an approximate annual savings of \$0.87M annually. By using health impact metrics based on reducing NO2 compounds in the air, Nui et al. estimated the annual health benefits to be between \$0.24 and \$3.27M.

The installation cost of a green roof is typically 27% higher than that of a conventional roof but lasts 40 years on average as compared to 20 years for a conventional roof (Nui et al., 2010). By including the savings in stormwater management fees and reduced energy consumption in their calculations, Nui et al. concluded a break-even point of seven years.

2.2.2 Locating Roof Greening Opportunities

Timothy Carter and Rhett Jackson (2007) found that there is high potential to manage a significant volume of stormwater using existing rooftops, however detailed spatial analysis needs to be performed in order to direct management efforts. They estimated reductions in stormwater runoff from 3 roof greening scenarios in the Tanyard

Branch watershed in Athens GA, which encompasses much of the University of Georgia and the urban center of Athens. Three roof greening scenarios were examined: no green roof implementation, all roofs greened, and only flat roofs greened. They performed a detailed inventory of impervious surfaces including roof type (sloped or flat) and employed the Soil Conservation Service curve number method to model the infiltration and runoff of the site for a variety of different rain events and spatial scales. Spatial scales included the total watershed, sub-watersheds, by zoning, and at the parcel level. Residential areas ranged from 35% to 45% total impervious area (TIA), and commercial areas ranged from 54% and 78% TIA in the Tanyard Branch Watershed. While rooftops accounted for almost 30% of all impervious surfaces, they constituted less proportion of TIA in commercial areas versus residential areas. The majority of impervious surface in commercial areas was composed of parking lots which typically contain the largest concentrations of vehicular pollutants and therefore pose a particular risk to receiving water bodies. Approximately half of the roofs in the watershed were flat and flat roofs were concentrated in the commercial areas while residential zones included few flat roofs. However, rooftops in commercial areas were more directly connected to the storm sewer system and flat roofs are typically easier to green

Carter and Jackson discovered that when flat roofs were disaggregated from the spatial data, a clear hierarchy for green roof implementation emerges at the zoning level. They found that the greening of only flat roofs would result in an 18.9% reduction in runoff volume for 60% of all rain events. While a comprehensive roof greening scenario resulted in the highest volume reduction, they suggested that low impact development modeling focused on residential applications shows little opportunity for practical green roof implementation. This is partly because residential roofs are not as connected to other impervious surfaces. Instead, many yards already provide a degree of stormwater management but also provide opportunities for other types of BMPs, such as rain gardens, which may be more practical for homeowners to install. Carter and Jackson concluded that areas of commercial, industrial and institutional zoning (and consequently contain most large flat-roof buildings) should be targeted for retrofit installations.

2.3 Green Roof Policy

The additional costs associated with green roof construction and negative perceptions (as some building owners may believe their roof to be more leak prone due to greening), impede the adoption of green roof technology (Carter and Fowler, 2008). Carter and Fowler argued that because green roofs provide many public benefits that are not necessarily realized by the party bearing the cost of installation, public intervention is justified. In conducting a public survey to evaluate existing international and North American green roof policies at the federal, municipal, and community levels, Carter and Fowler (2008) found that green roof policies can be categorized as direct and indirect regulation, direct and indirect financial incentives and funding of demonstration or research projects. Provisions in the Clean Water Act (CWA) that require approval for National Pollutant Discharge Elimination System (NPDES) permits include provisions to treat stormwater runoff using best management practices such as green roofs (Carter and Fowler, 2008). More detailed green roof policies are found at local levels of government and can be considered in terms of either regulation or incentives. Direct regulation involves enacting technology or performance standards where municipal governments either directly require green roof implementation or require buildings to manage a specific amount of stormwater or heat reflection through building code requirements. Indirect regulation involves using a market based approach where local governments charge fees for stormwater management based on amount of impervious area and give fee credits or density bonuses for green roof installations. Financial incentives provided for by local governments can also be direct or indirect. The most direct financial incentive is the funding of green roof implementation through subsidies which Carter and Fowler argued help overcome the financial barriers of adopting the new technology. Each of these approaches has distinct advantages and disadvantages. “Direct financial incentives in the form of subsidies have the advantage of providing building owners compensation for initial construction costs”, however “jurisdictions must have adequate funding sources to provide this subsidy” (Carter and Fowler, 2008). They also pointed out that mandating green roofs through building codes may provide a high level of assurance that the infrastructure will actually be built, but is likely to be politically unpopular. “Indirect financial incentives and performance standards have the advantage of being voluntary, favoring those owners who can install green roofs in a cost effective manner based on their site conditions”, however “a disadvantage is that it is difficult to guarantee green roofs will be installed” (Carter and Fowler, 2008).

2.4 Urban Ecology and Green Roofs

Urban development is typically viewed only as a process of wildlife and plant habitat destruction. Indeed urban development has had profound negative impacts on natural ecosystems (Alberti et al., 2003). But urban development also results in the creation of novel ecosystems. The habitats that result from urban development are composed of mostly hard surfaces and poor soils, making them more physically analogous to certain natural habitats such as rocky outcroppings, limestone barrens, or dry grasslands (Lundholm, 2006). Urban plant communities contain mixes of native and non-native species that interact in anthropogenically-driven successions (Alberti et al., 2003). They are usually marked by low stability and low species diversity but can equilibrate into stable communities over time (Alberti et al., 2003). Some urban habitats, such as post-industrial brownfield sites in England, are characterized by soils that share similarly dry and nutrient poor conditions with indigenous habitats that contain very diverse plant communities (Gedge and Kadas, 2004). These urban “waste” sites are essentially man-made habitat templates that have been colonized by plants and animals native to analogous habitats (Gedge and Kadas, 2004). Brownfield sites are now among the most species rich habitats left in the U.K., as they have provided refugia for many species (including some that are very rare) from surrounding plant and animal communities (Kadas, 2006) that have been replaced by agricultural expansion (Gedge and Kadas, 2004).

Green roofs are perhaps the oldest form of living architecture resulting from merging the built environment with living ecosystems (Lundholm and Peck, 2008). Green roofs can serve as components of an urban ecological patchwork, however ecologists have paid little attention to green roofs as ecosystems (Lundholm and Peck, 2008). The habitats that extensive green roofs provide are harsh environments of extreme temperature fluctuations, periods of prolonged drought, and periods of water inundation (Lundholm, 2006). These extremes in microclimate and hydrology mean that arrested succession communities such as grassland, tall herb, succulents, moss mats and bare ground are more likely to be established on green roofs (English Nature, 2003). Plants that can tolerate these extreme conditions can be found growing in naturally analogous habitats in the wild (Lundholm, 2006) and these habitats have been used as models for green roof design. Designers of a pioneer green roof project in Nashville, Tennessee, used cedar glades as a habitat template to choose an appropriate plant community (Shriner, 2003). Cedar glades are a rare habitat type occurring near the Nashville area that include characteristically thin soils over rock that stay waterlogged through much of the winter and are extremely dry during the summer (Shriner, 2003). This choice in habitat templates led the designers to plant species of rare plants indigenous to cedar glades such as the federally endangered Tennessee coneflower and purple prairie grass. This reinforces existing evidence regarding other climates that green roofs can provide habitat for specialist rare and endangered species (English Nature, 2003; Gedge and Kadas, 2005; Brenneisen, 2006) for a humid subtropical climate.

In addition to plants, green roofs can provide habitat for a range of birds and invertebrates (Brenneisen, 2003; English Nature, 2003; Gedge, 2003; Coffman and Davis, 2005, and Schrader and Boning, 2006). A 2004 survey of fauna on the Ford Motor Company green roof in Dearborn, Michigan, planted with sedum mats in 2002, revealed that 29 families of winged insects, 7 spider species and 2 species of birds were using the roof (Coffman and Davis, 2005). The Moos Water Filtration Plant green roofs in Zurich, Switzerland, host 254 beetle species and 78 species of spider (Brenneisen, 2006). Many avian species including some that are rare or endangered have been recorded using green roofs for foraging, nesting and other activities (Brenneisen, 2003). Most of the species of birds found using green roofs in Switzerland naturally occur in open landscapes such as high mountain areas, river banks, or in grasslands while species that are very commonly found in urban areas were largely absent (Brenneisen, 2003). Green roofs in Swiss suburban areas were used by birds much less frequently than the urban green roofs, presumably because of availability of green space in these areas (Brenneisen, 2003). Brenneisen advocates using microhabitats, created through topographic variation in substrate depth, and the incorporation of native soils and substrates on individual green roofs to increase biodiversity to near natural habitat levels (Brenneisen, 2003; Brenneisen, 2006). Some green roofs in London have been designed for conservation strategies specifically targeting the Black Redstart, an endangered species of bird that requires habitats similar to urban brownfield sites for reproduction (Gedge, 2003).

It is important to note, however, that green roofs have an extremely limited ability to replicate existing habitats due to both physical limitations and limitations of scale (English Nature, 2003; Brenneisen, 2006). Green roofs cannot host certain species due to either their inability to reach the elevated substrates or adapt to the harsh environments present there (Brenneisen, 2006). Furthermore it may be impossible to recreate or maintain exact soil and hydrological conditions in part because the roof is not in contact with natural groundwater (English Nature, 2003). Limitations of scale include time; a habitat may not be re-creatable within a feasible timescale, and size; a patchwork of green roofs, small relative to the habitat being surrogated, is unlikely to make a significant impact in terms of conservation (English Nature, 2003). English Nature Report Number 498 (2003) states that only when a certain threshold of green roof area is reached will conservation and other environmental benefits likely to become apparent.

2.5 Biodiversity and Succession on Green Roofs

Even though interest in modern green roofs initially arose from their potential as biodiverse urban habitats in Germany in the 1960's (Gedge and Kadas, 2005), existing literature largely focus on the more direct financial benefits of stormwater management and energy conservation. Less is known about what green roofs can achieve in terms of biodiversity (Kadas, 2006). In a regional context, green roofs have the potential to provide new plant and wildlife habitat in areas where it is lacking, act as habitat corridors by facilitating species movement and dispersal and act as refugia for rare species (English Nature, 2003, Kadas, 2006). Relatively little is known about the levels of biodiversity that can be achieved on individual green roofs. This is especially true in the United States, which lags behind European nations in green roof research (Dvorak and Volder, 2010). North America contains many distinct ecoregions and climates which compounds research needs because findings from one study may not be applicable to other areas (Dvorak and Volder, 2010). Dvorak and Volder (2010) stated that there is great need for green roof biodiversity research in North America, especially in ecoregions where biodiversity is in decline.

Approaches to increasing green roof biodiversity in Europe have included spontaneous plant colonization and seed bank sampling. The city of Basel, Switzerland, as part of their biodiversity strategy, now requires green roofs on all flat roofs and those over 500 square meters must be constructed with natural soils. Brown roofs are green roofs that have been designed to emulate urban brownfield habitats (Bates, Mackay, Greswell and Sadler, 2009) which are defined as formerly developed land (Brenneisen, 2006). One of the strategies used in creating brown roofs is to sample brownfield seed banks and allow natural succession to occur (Brenneisen, 2006; Bates et al., 2009). Allowing plants to spontaneously colonize green roof soil media is an approach that has also gained attention from those interested in the biodiversity potential of green roofs in England, yet this is something that has happened more by default rather than design (English Nature, 2003). Most green roof experiments represented in the literature have involved the removal of spontaneously colonizing plant species so ecological measurements of naturally forming plant communities on green roofs has largely been limited to examining existing installations that have been allowed to accumulate species. Therefore there is a distinct lack of knowledge regarding plant community succession on green roofs. There is evidence, however, that green roofs can accommodate extremely diverse plant communities. Four green roofs atop the Moos Water Filtration Plant in Zurich, Switzerland were built in 1914 using native soil and left undisturbed thereafter (Landolt, 2001). At 90 years old, these green roofs are home to 175 plant species including 9 species of orchids and others that are now rare or extinct in the area. The roofs contain more or less entire plant communities known from ground level wet meadow habitats and reflect the species richness of the area as a farming region in the early 20th century (Landolt, 2001) before agricultural intensification (Brenneisen, 2006).

Biological diversity is not only defined as the variety but also the abundance of species in a defined unit of study (Magurran, 2003). In addition to *species richness*, or the number of species present in the unit of study (Magurran, 2003), a measurement of diversity must also include a measurement of *evenness* (Simpson, 1949), which describes the variability in abundance of those species (Magurran, 2003). Smith and Wilson (1996) explain that "a community in which each species present is equally abundant has high evenness; a community in which the species differ widely in abundance has low evenness." (Smith and Wilson, 1996, pg. 70). Magurran (2003) suggests that in studies of biodiversity, species abundance patterns deserve equal if not greater attention than species richness.

Studies regarding the biodiversity of green roofs in terms of species richness are represented in the literature, but this work focuses almost exclusively on green roof fauna rather than spontaneous green roof flora (Dunnett, Nagase and Hallam, 2008) (e.g. Gedge, 2003; Gedge and Kadas, 2004; Coffman and Davis, 2005; Brenneisen, 2006; Schrader and Boning, 2006). There have been few studies that involve manipulating plant diversity on green roofs (Cook-Patton and Bauerle, 2012) and attempts to quantify biodiversity on green roofs that incorporate evenness are very rare. Not surprisingly, the investigations that have explored the ecological dynamics of green roofs have revealed dynamics similar to those in naturally occurring ecosystems (Cook-Patton and Bauerle, 2012).

Nigel Dunnett (Dunnett and Allison, 2004; Dunnett et al., 2008) conducted a study that manipulated and quantifyied the ecological characteristics of an emerging plant community on green roofs in Sheffield England from Spring 2001 through winter of 2006. Fifteen species of native perennial grasses and forbs were planted in two soil depth treatments of 100 mm (3.937 in) and 200 mm (7.874 in). Ecological measurements of species richness, abundance in terms of mean number of individuals per test plot, and Shannon-Weiner diversity, a measurement that includes information on species richness and evenness (Magurran, 2003) were conducted separately for planted species and spontaneously colonizing species. Though spontaneously colonizing plants were included in the study, Dunnett et al. (2008) stated that all plants were clipped to a height of 100 mm at the end of the first and second growing seasons and "weeds" were identified and removed at this time. They also state that the numbers of colonizing species were recorded in 2004 and 2005 but were removed afterwards, thereby affecting succession. The investigators found that deeper substrates resulted in the greatest survival and abundance (and therefore greater diversity) of the 15 planted species. The deeper substrates also resulted in greater biomass (productivity) of both planted and colonizing species and overall cover. Although diversity of planted species was consistently higher for deeper substrates, diversity between the two treatments was not significantly different and there was an overall decreasing trend in diversity of both treatments which appeared to be converging. The shallower substrates, however, supported the greatest species richness, abundance and Shannon-Weiner diversity of colonizing species, bare ground and moss. Altogether, 35 species of colonizing plants were recorded. Dunnett et al. (2008) described the colonizing vegetation as a mix of native and exotic species typical of cosmopolitan urban plant communities but described them as ruderal weeds of wasteland and agricultural disturbance. Dunnett et al. (2008) also say that even though the shallow substrates exhibited greater species richness and diversity, the plant communities exhibited low evenness, as a small number of species accounted for the majority of individuals, biomass and percentage cover. Because of this, Dunnett et al. (2008) warn that relying on spontaneous colonization may lead to the development of low diversity systems where one aggressive species dominates.

Other studies that involve manipulating plant diversity on green roofs have focused on quantifying the effects of diversified plantings on specific green roof functions, such as stormwater capture and summer roof cooling, and not the ecological characteristics of the plant community (Dunnett et al., 2008) (e.g. Kolb and Schwartz, 1986; Dunnett, Nagase, Booth and Grime, 2008; Lundholm, MacIvor, MacDougall and Ranalli, 2010; MacIvor, Ranalli and Lundholm, 2011). All of these studies concluded that greater plant diversity on green roofs offers opportunities for optimizing green roof function, but also that diversity alone does not necessarily optimize function as different plants effect functions in different ways. One study examined the effects of diversity on plant survival and the investigators concluded that a diverse mixture of sedums, forbs and grasses was advantageous under drought conditions (Nagase and Dunnett, 2010). However, almost all American green roof experiments have relied on supplemental irrigation for plant establishment and survival.

Evidence of biodiversity on green roofs increasing with succession can be found in the diversity of very small invertebrates and microorganisms living in their soil media (Schrader and Boning, 2006). The soil media of extensive green roofs is typically comprised of mostly mineral soils with very little organic matter (Lundholm, 2006) but the constitution of these soils change over time (Schrader and Boning, 2006). In a German study that compared the growing medium of older green roofs between 8 and 12 years old, and newer green roofs between 3 and 4 years old, it was found that natural soil formation was occurring and microbial activity increasing. (Schrader and Boning, 2006). The researchers looked at the diversity of Collembolan species, a large genus of soil dwelling arthropods (Meyer, 2006) as an indicator of soil conditions. They discovered that relatively new green roofs play host to ubiquitist species of Collembolans while older green roofs host more specialist Collembolan species. This seems to demonstrate that green roof soils develop over time and trend towards more natural soil conditions. A similar study examined changes in collembolan communities during primary succession of afforested mining sites (Dunger, Schultz, Zimdars, and Hohberg, 2004). These afforestation efforts dealt with sterile growing substrates similar to those of extensive green roofs. The findings of the study showed that even after fifty years there were marked differences between collembolan communities in adjacent reference woodlands and the afforested sites (Dunger et al., 2004). These findings imply that the growing media of green roofs, like the reclaimed mining spoils, may develop characteristics of native soils over time, but may never be indistinguishable. Regardless, Schrader and Boning (2006) discovered that over time, green roof soil media becomes more habitable to a broader range of invertebrates and microbes and therefore more diverse.

2.6 Use of Non-Succulent Native Plants on Green Roofs

Extensive lists of plants that have proven successful for green roof use have been developed for European applications whereas a comprehensive literature review of successful plants for use on North American green roofs revealed only 40 succulent, and 94 non-succulent herbaceous species that have exhibited success (Dvorak and Volder, 2010), though the majority of these were established with irrigation. Furthermore, these findings are restricted to only a few ecoregions and climates (Dvorak and Volder, 2010). Research regarding appropriate plants for use on extensive green roofs has focused on their ability to survive the harsh growing conditions. Succulent plants, such as members of the sedum genus, have commonly been recommended and used as green roof vegetation because of their extreme drought tolerance (Butler et al., 2012, Getter and Rowe, 2006; Monterusso et al., 2005; White and Snodgrass, 2003). Indeed, there seems to be a consensus among researchers that succulent plants exhibit better rates of survival than non-succulent herbaceous species, especially in the absence of irrigation (Dvorak and Volder, 2010). However, these species are not native to many regions in the United States and local plant communities, which likely contain thousands of potentially useful species, have largely been overlooked (Sutton et al., 2012). A blanket approach of exclusively using succulents for green roof applications also presents some problems. According to Emilsson (2008), green roofs dominated by succulents have limited value for plant biodiversity because fewer species spontaneously colonize them. Nagase and Dunnett (2010) argued that even though they found sedums to be superior to forbs and grasses in terms of drought tolerance, ecological theory suggests that a highly species-rich plant community might be more resistant to severe environmental stress. Overuse of sedum monocultures on green roofs may lead to future problems with insects and disease, especially considering a scenario where tens of thousands of square feet of urban roof tops are planted (Sutton, 2008). It is known that sedum species are susceptible to insect and fungal problems such as mold and root rot (Sutton, 2008), which may prove especially suboptimal for use in hot and humid climates (Livingston, Miller and Lohr, 2004). Furthermore, the long term sustainability of green roof projects will depend on biodiversity and natural nutrient cycling to replace the need for fertilizers to restore nutrients to spent growing media (Sutton, 2008). There is also evidence that more diverse groupings of plant taxa offer greater benefits in terms of stormwater retention, temperature regulation and absorbance of solar radiation (e.g. Dunnett and Nagase, 2008; McIvor and Lundholm, 2011). McIvor and Lundholm (2011), for example, found that some species locally indigenous to Hallifax, Nova Scotia, exhibited improved performance over commonly used sedum and grass species in regard to these functions.

There has been a great deal of interest in growing native, non-succulent plants on green roofs (Butler et al., 2012; Getter and Rowe, 2006). Dvorak and Volder (2010) suggested that there are many native or introduced non-succulent herbaceous plants that can be effectively used on extensive green roofs, but also suggested that many may require deeper substrates and irrigation. There seems to be a high degree of uncertainty throughout the literature as to whether non-succulent herbaceous plants can survive on green roofs without irrigation. This seems unlikely, however, considering non-succulent plants have been observed spontaneously colonizing tar rooftops with no substantial growing media in hot and sunny climates (e.g. Shriner, 2003). In some regions of the United States, there has been particular interest in using green roofs to recreate prairie habitats (Getter and Rowe, 2006; Dvorak and Volder, 2010). However, much of the published research seems to be negative or inconclusive regarding the efficacy of native prairie species for extensive green roof use in the U.S. (Sutton, 2008). A frequently cited study conducted by Monterusso et al. (2005) concluded that native prairie plants from Michigan were unable to thrive on green roofs without irrigation. However, several authors have questioned this conclusion. Some argue that many of the plants selected for this study do not seem to follow an appropriate habitat template. Sutton (2008) pointed out that the plants used in the study were indigenous to tallgrass prairies and argued that the shallow substrates will not accommodate their deep rooting habits. Getter and Rowe (2006) suggested that species from shortgrass prairies are probably better suited for extensive green roof use, however shortgrass prairie systems are native to the more arid regions found farther west and southwest in the United States and therefore may defeat the intentions of using locally native prairie species in other regions. Sutton (2008) also argued that prairie species need at least 4 years to form the dense root network that would be needed to survive without irrigation whereas the researchers ceased irrigation after two years. Snodgrass and Snodgrass (2006) questioned the ability of prairie species to thrive on green roofs in general due to these possible depth requirements and further suggested that replicating a prairie ecosystem on green roofs may be impossible because many prairie species have evolved in very particular soils of an exact microbial and nutrient balance. Therefore much of the plant taxa that make up a prairie may not be reasonably accommodated. However, Sutton et al. (2012) reminds us that green roofs and naturally occurring grassland environments share very similar conditions such as intense wind and sun exposure, low moisture, high evapotranspiration rates and frequent drought conditions. Jeremy Lundholm, plant ecologist and associate professor of biology at St. Mary's University, Nova Scotia, suggested that rock outcrop communities or really dry prairie vegetation over bedrock (cedar glades, alvars, limestone pavement) might be better habitat template to follow (J. Lundholm, personal communication, February 18, 2011). Many of these plants may also be succulents (White and Snodgrass, 2003).

Besides survivability, there are other challenges to using non-succulent natives for green roof applications. Many of these issues revolve around aesthetic goals. White and Snodgrass (2003) pointed out that practitioners wishing to install green roofs will likely desire a long flowering season (White and Snodgrass, 2003). According to green roof experiments using non-succulent native plants, not only is mortality high but visual appearance declines in terms of perceived plant health as these plants enter dormancy during periods of drought (Monterusso et al., 2005) and often do not quickly resume growth (White and Snodgrass, 2003). Furthermore, dried plant tissues from grasses and other native perennials may present a fire hazard whereas succulents store water in their stems and leaves, and act as a fire retardant (White and Snodgrass, 2003). However, English Nature report number 498 refutes the notion that green roof thatch presents a significant fire hazard (English Nature, 2003) There is some evidence that succulents can aid in the survival of non-succulents when used in combination. A study conducted by Butler and Orians (2011) revealed that sedums can have a positive effect on non-succulent herbaceous plant performance during hot and dry weather but had a negative effect during more favorable conditions.

2.7 Soil Media and Irrigation in Similar Studies

Other than regional climatic conditions, irrigation and the depth and composition of soil media appear to be the most influential variables influencing plant performance on green roofs (Dvorak and Volder, 2010). While several studies have focused on the effects of soil media composition on various aspects of green roof performance (e.g. Getter and Rowe, 2006; Monterusso et al., 2005; Simmons et al., 2008; Rowe, Monterusso and Rugh, 2006), research regarding modification of soil media in order to enhance the survival of native, non-succulent herbaceous vegetation is less common. Gedge and Kadas (2004), Dunnett (2006) and Brenneisen (2006) have advocated the inclusion of native soil from a specific habitat to be emulated on a green roof with the expectation that the seed bank, as well as other organisms would be sampled. Dunnett (2006) suggested that the native soil contains microflora such as a range of mycorrhizal fungi that may aid in plant establishment. However, it was also noted that the ecological characteristics of the soil are changed once it is disturbed and that ruderal or "weedy" species present in the seed bank are likely to benefit the most from this. The results of an experiment conducted by Sutton (2008) revealed that the combined addition of hydro-absorbent polymer gel and native prairie soil as a microbial inoculant enhanced plant vigor in a planting of prairie grasses, sedges and forbs in Lincoln, Nebraska. This study implemented regular irrigation and so it is not known how the additions would have effected survival in prolonged drought conditions.

Several high-profile prairie green roofs have been implemented and studied in the United States, such as the California Academy of Sciences (Kephart, 2009), Church of Jesus Christ of Latter Day Saints Conference Center in Salt Lake City, Utah (Butler et al., 2012), The Neuhoff Meat Packing Plant building in Nashville, TN (Shriner, 2003) and the Chicago City Hall Green Roof Pilot Project (Dvorak and Carroll, 2008). However all of these green roofs implement regular irrigation and the LDS Conference Center and the Chicago City Hall green roofs implement soil media depths beyond that of extensive systems.

Very little research has examined the efficacy of using native vegetation in the climates of the southern United States (Dvorak and Volder, 2012). A study conducted in central Florida's humid subtropical climate successfully established a mixture of exotic and native succulent and herbaceous vegetation on green roofs using irrigation (Wanielista and Hardin, 2006), and another conducted in the tropical climate of south Florida, established a similar mixture in 15.24 cm (6 in) of soil media without irrigation (Livingston et al., 2004). Simmons et al. (2008) conducted a study in central Texas and found that a mixture of native grasses and forbs effectively achieved stormwater retention and roof cooling but established the plants with irrigation and did not quantify plant performance. As of December, 2012, only Dvorak and Volder (2012) had published a study on establishing native vegetation on unirrigated green roofs in a subtropical climate. They successfully established 10 species of native and exotic plants in 11.4 cm (4.48 in) of soil media without irrigation in south-central Texas. However all of the species used were succulent except *Nassella tenuissima*, which is a native grass.

CHAPTER III

MATERIALS AND METHODS

3.1 Research Design

The research design is a paired green roof experiment that compares 5 control replicates (green roofs constructed with commercially available soil media only), and 5 treatment replicates (green roofs constructed with a mixture of commercially available soil media and native soil). All roofs were planted with native species as plugs and seeds and colonization was allowed to proceed freely. The independent variable was the incorporation of native prairie soil into commercially available green roof soil media. The dependent variables were survival of plug species, success of seeded species, percent cover of all species present, and diversity in terms of species richness and evenness. The null hypothesis was that the introduction of native prairie soil does not influence the establishment and survival of native herbaceous plants on an extensive green roof system. The alternative hypothesis is that the introduction of native prairie soil does affect the establishment and survival of native herbaceous plants on an extensive green roof system. The experiment began in May, 2011 and data collection was concluded September, 2012.

3.2 Study Site

The primary study site was the H. H. Laveck Animal Research Center on the campus of Mississippi State University which is located approximately 33° 25’ 25” N and 88° 47’ 32” W, with an approximate elevation of 327 feet above sea level. The research area is predominantly surrounded by cow pastures and agricultural test plots. The University Greenhouses at Dorman Hall were also used to initially transfer and house plug species. The climate of Mississippi is generally considered to be humid subtropical, which is characterized by mild winters without extended periods of temperatures below freezing and long, hot summers with no routinely recurring wet or dry season (National Oceanic and Atmospheric Administration, 2005). According to NOAA, Mississippi’s wettest period in terms of total precipitation is November through June, with March and April receiving the greatest frequency of rain events. However, due to contrasting influences of topographic features and air currents, the average conditions are rarely present for extended periods. Instead, winter and spring seasons can be warm or cold, wet or dry, and summers can have extended periods of drought or frequent rain events (National Oceanic and Atmospheric Administration, 2005). Between 1971 and 2000, the hottest months at Mississippi State University have been July and August with average high temperatures just above 90°, and the coldest month has been January with an average low temperature of 31.5°. (Table 3.1) Between 1971 and 2000, the driest months at Mississippi State University have been August through October with an average monthly rainfall of 3.39 inches, and the wettest month has been March with 6.07 inches. (Table 3.2)

Table 3.1 Monthly Average Temperatures for Mississippi State University in °F, 1971–2000

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
Max	51.9	57.2	65.8	73.9	81.3	88.1	91.3	90.8	85.3	75.6	64.6	55.3	73.4
Mean	41.7	46.1	54.2	61.8	70.2	77.5	81.0	79.8	74.0	63.0	53.4	45.0	62.3
Min	31.5	34.9	42.5	49.7	59.0	66.8	70.6	68.8	62.6	50.4	42.2	34.6	51.1

(National Oceanic and Atmospheric Administration, 2005)

Table 3.2 Monthly Average Precipitation for Mississippi State University in inches, 1971 – 2000

Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
5.70	4.85	6.07	5.62	4.88	4.03	4.35	3.33	3.48	3.35	4.66	5.13	55.45

(National Oceanic and Atmospheric Administration, 2005)

3.3 Green Roof Platforms

Ten experimental, simulated green roof platforms were constructed in June, 2011 using pressure treated pine lumber. Each platform is 4 ft (1.219 m) by 4 ft (1.219 m), providing 16 ft2 (1.486 m2) of surface area. Each platform is 8 in. (20.32 cm) deep, accommodating 6 in. (15.24 cm) of growing media, a drainage layer, a waterproof membrane and freeboard. The platforms had a 2% slope to simulate a typical flat roof. An opening on the bottom of the lowest side of each platform allowed for excess water to freely drain from the system. Each platform is elevated 6 ft (1.8 m) above the ground and rests on a frame of 4/4 posts. The frame is open, allowing for air circulation on all sides. (fig. 3.1)

Figure 3.1 Illustration of Simulated Green Roof Platform.

A waterproof membrane was constructed by covering the interior surfaces of the platforms with a double layer of fully-adhered, SBS-modified Bitumen (Sopralene Flam GR, Soprema, Wadsworth, OH), simulating the waterproof membrane of a typical flat roof used for commercial applications.

A cup-style, or dimpled drainage layer (J-Drain® GRS, JDR Enterprises Inc., Alpharetta, GA) (JDR Enterprises, 2003) was used to line the bottom of each platform and rests atop the waterproof membrane. A cup-style drainage layer was chosen in order to add extra water retention and extend the availability of moisture to the soil media. In order from top to bottom, the drainage layer is composed of a root-resistant filter fabric, a plastic dimpled drainage core and a bottom protection fabric. The dimpled drainage core retains 1.1 gallons of water per 10 ft^2 upon saturation. The drainage layer weighs .44 lbs/ft^2 and its overall thickness is 1 in.

Six inches of growing media were placed atop the drainage layer. The growing media that was used is a commercially-available media (ERTHHydrocks Lightweight Soil Media-Extensive, ERTH Products, Peachtree City, GA) for the experimentally controlled platforms, and a mixture of the same commercially available media and locally-native prairie soil for the experimentally-treated platforms. ERTHHydrocks Lightweight Soil Media-Extensive is composed of 50-80% 3/8”-3/16” hydrocks rotary kiln expanded clay particles, 5-20% nutrient grade compost and 5-30% USGA (U.S. Golf Association) sand.

3.4 Treatment

Two different growing media compositions were examined and each composition was replicated five times on simulated green roof platforms. The experimentally-controlled media consisted of commercially available green roof media only (ERTHHydrocks Lightweight Soil Media-Extensive). The experimentally treated media included a mixture of native soil. Locally-native prairie soil comprised 16.66% by volume of the growing media used for the experimental treatment. The soil was extracted as hand-dug plugs, approximately 4 to 6 in. (10 – 15 cm) deep, from two nearby relict prairie sites. The sites were Osborn Prairie (33° 30’ 37” N, 88° 44’ 14” W, 300 ft. above sea level) near Osborn, Mississippi, and the Mississippi State University Dairy Unit (33° 23’ 22” N, 88° 44’ 26” W, 298 ft. above sea level) near Sessums, Mississippi. A soil analysis was performed on a sample taken from a volume of combined prairie soil samples and revealed a composition of 70.75% silt, 26.75% sand and 1.25% clay with a pH of 7.8. A complete comparison of the soil analyses of the native soil and the manufactured media is available in Table 3.3. Large organic debris such as roots and twigs were removed by hand. The soil was allowed to dry naturally and aggregate clumps were broken down into particles ranging in size from 3/8 in. to a fine dust. The native soil and commercial media were mixed at a ratio of 1:5 until the components were evenly combined. The surface of the final mixture was dusted with pure native soil in situ.

Table 3.3 Soil Analyses of Commercial Media and Native Soil.

Quality	**Green Roof Media**	**Native Soil**
pH	5.7	7.8
P (lbs/acre)	1248	49
K (lbs/acre)	464	366
Ca (lbs/acre)	1561	9590
Mg (lbs/acre)	298	324
S (lbs/acre)	392	337
Zn (lbs/acre)	28.1	18.6
Na (lbs/acre)	50	81
% Organic Matter	2.72	2.34
% Clay	1.25	2.50
% Silt	7.50	70.75
% Sand	91.25	26.75
Texture	Sand	Sandy Loam

3.5 Plant Establishment

Plants were introduced into the simulated green roof platforms as both potted specimens (plugs) and seeds. Five species of locally-native prairie plants, Sideoats Grama (Bouteloua curtipendula), Blue Mistflower (Conoclinium coelestinum), Purple Coneflower (Echinacea purpurea), Yellow Coneflower (Ratibida pinnata), and Heath

Aster “Snow Flurry” (Symphyotrichum ericoides), were ordered from North Creek Nurseries (Oxford, Pennsylvania) as 2 in. by 5 in. plugs in December, 2010. These species were chosen based on being locally occurring prairie species and availability. The plugs were transferred into 1 gal pots of either control media (ERTHHydrocks Lightweight Soil Media-Extensive only) or treatment media (ERTHHydrocks Lightweight Soil Media-Extensive with native soil added) on May 4, 2011. The plants were housed in the Dorman Greenhouses until June 7, 2011, when they were transferred to the study site. At the study site, the plants were placed directly on the ground, grouped according to species and treatment until being planted on the simulated green roof platforms in September. Each potted plant was randomly assigned to a unique location on a simulated green roof platform corresponding to the appropriate treatment. The result was each platform received 16 randomly assigned plants that were planted 12 in. (.3048 m) on center. (Figs. 3.2, 3.3) Planting began Sep. 8, 2011, and was concluded Sep. 23, 2011. All scientific names of plants are from USDA, NRCS (2013), retrieved from http://plants.usda.gov, March, 2012.

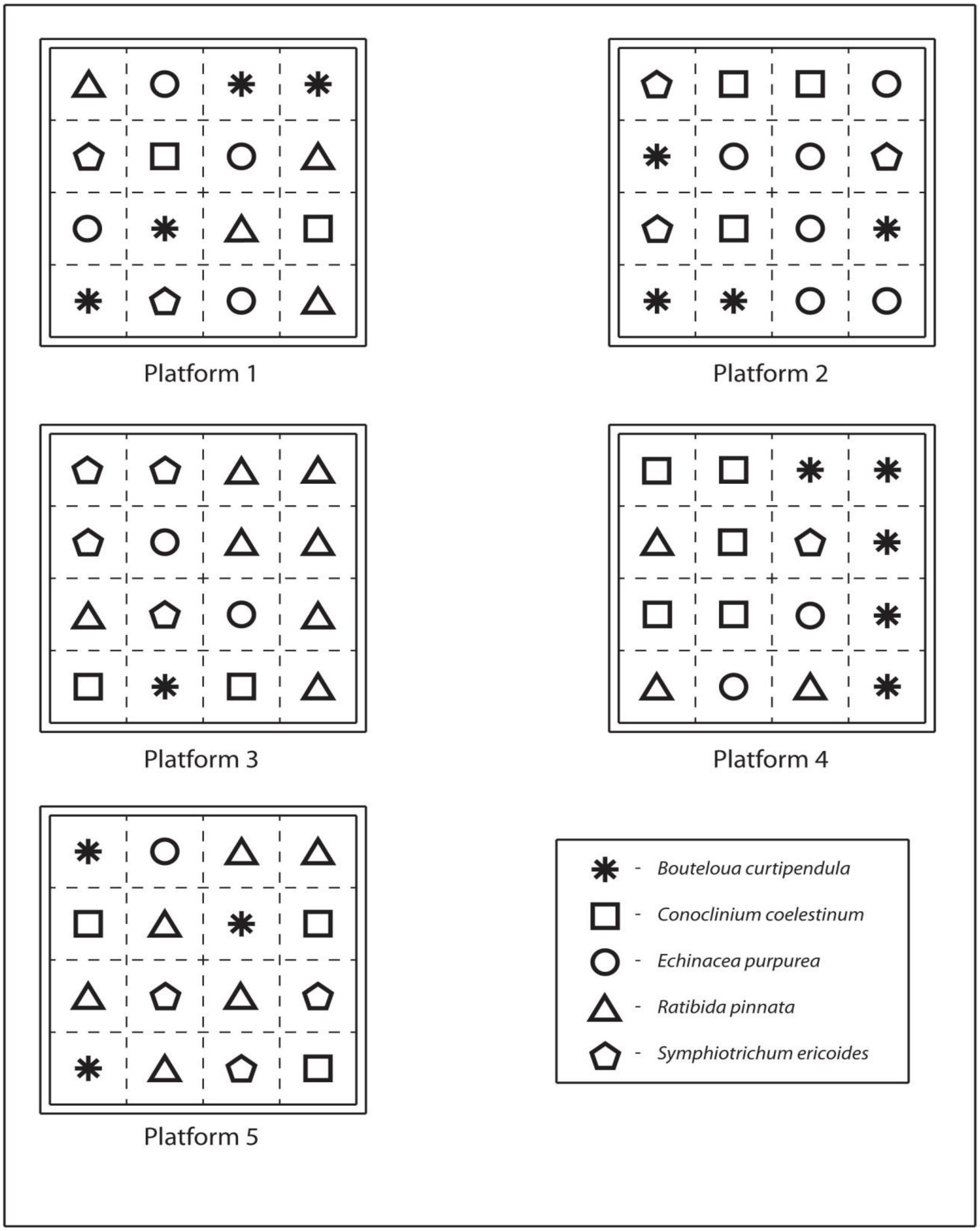

Figure 3.2 Randomized Planting Arrangement for Control Roofs.

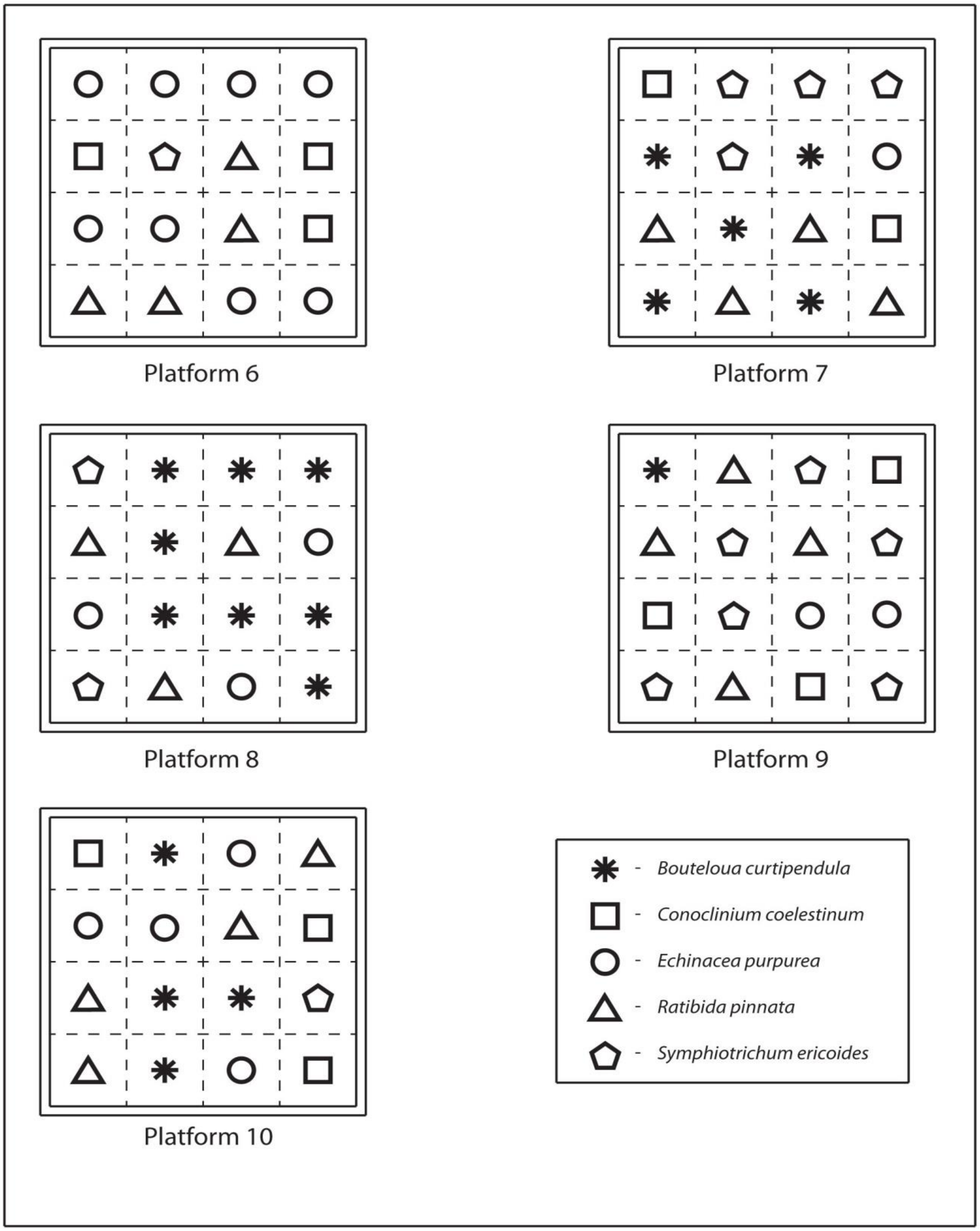

Figure 3.3 Randomized Planting Arrangement for Treatment Roofs.

Through the months of May and June the potted plants were watered until saturation once every two days. During July, the plants were watered once every three days, and during August and September they were watered once every four days. Each simulated green roof platform was watered until saturation upon completion of planting. All supplemental watering was terminated October 1, 2011, after which time the plants received natural rainfall only. (Fig. 3.4)

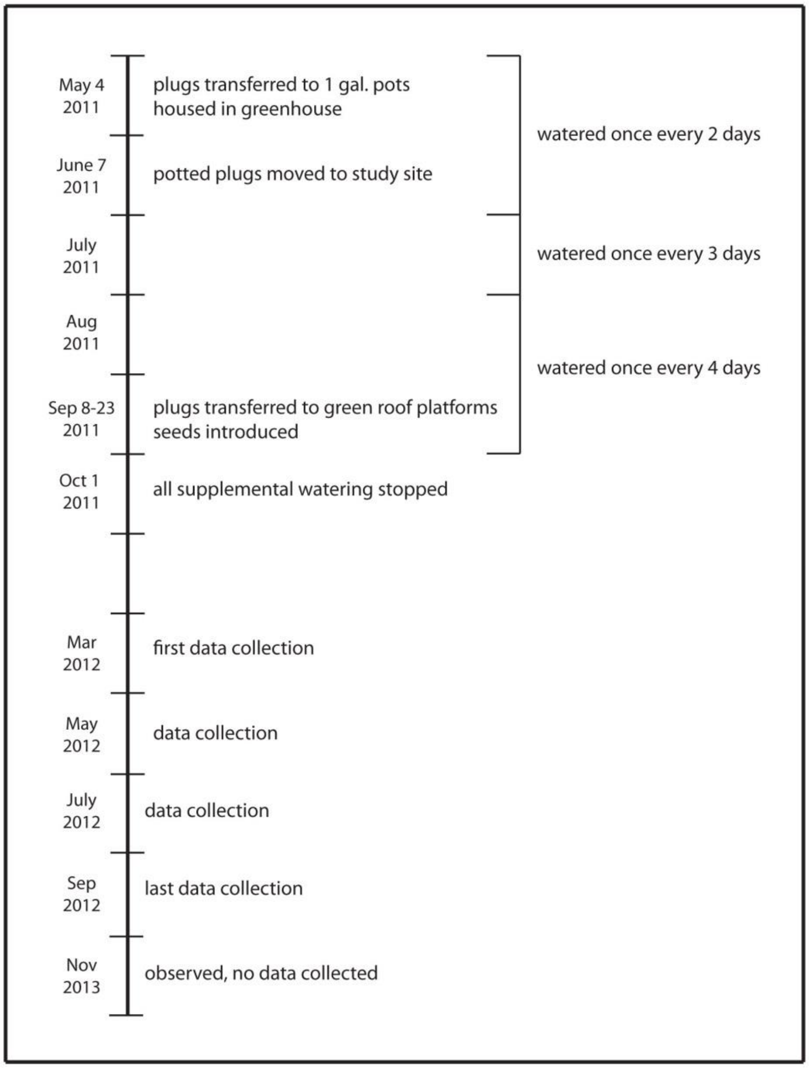

Figure 3.4 Timeline of Experiment.

Twenty-two species of locally native prairie plants were introduced to the platforms as seeds. (Tbl. 3.4) These species were chosen based on being locally occurring prairie species and availability. All of the seeds were purchased from Native American Seed, Junction Texas, except those for Helenium amarum, which were collected on-site by hand. The seeds were mixed with a small volume of sand in order to distribute them evenly by hand. Spontaneous colonization by extraneous species was allowed, and these species were not removed from the platforms or pots.

Table 3.4 Seeded Species and Seeding Rate.

Species	**Common Name**	**Ammount per 16 ft^2**
Andropogon virginicus	Broomsedge Bluestem	1.32 g
Agalinis heterophylla	Prairie False Foxglove	500 seeds
Asclepias tuberosa	Butterfly Milkweed	60 seeds
Asclepias viridis	Green Milkweed	60 seeds
Bouteloua curtipendula	Sideoats Grama	3 g
Chamaecrista fasciculata	Partridge Pea	3.45 g
Coreopsis lanceolata	Lanceleaf Coreopsis	3 g
Coreopsis tinctoria	Plains Coreopsis	1 g
Dalea candida	White Prairie Clover	2 g
Dalea purpurea	Purple Prairie Clover	2.5 g
Desmanthus illinoensis	Illinois Bundle Flower	5 g
Echinacea purpurea	Purple Coneflower	5.6 g
Eryngium yuccifolium	Rattlesnake Master	60 seeds
Helenium amarum	Sneezeweed	1 g
Monarda citriodora	Lemon Mint	1 g
Nothoscordum bivalve	Crow Poison	500 seeds
Oenothera speciosa	Evening Primrose	1 g
Penstemon tenuis	Gulf Coast Penstemon	500 seeds
Ratibida columnifera	Prairie Coneflower	2.8 g
Rudbeckia hirta	Black-Eyed Susan	.75 g
Schizachyrium scoparium	Little Bluestem	2.66 g
Solidago nemoralis	Gray Goldenrod	500-700 seeds

3.6 Data Collection

Data was collected in March, May, July, and September of 2012. Survival and health of plug species was estimated by visual inspection. An index of health based on appearance in which each plant received a score between 0 and 3 was used to estimate the survival and health of plug species. A score of 0 indicated the individual was dead or dormant and was assigned if there was no visible living tissue above the soil surface. A score of 1 was assigned if a major portion of the plant was dead or damaged, or if the individual was severely lacking normal growth. A score of 2 was assigned if the individual exhibited a normal amount of growth but had a moderate amount of damage or appeared to be extremely stressed. A score of 3 was assigned if the plant appeared to be very healthy with only minor or normal amounts of damage or stress.

Percent cover of plug species as well as all other present species was estimated visually. A portable grid was used to divide each platform into 16, 1 ft. (.3048 m) x 1 ft. (.3048 m) cells for ease of visual estimation. The identity of spontaneously colonizing species was determined whenever possible and it was noted whenever any plant was flowering or had flowered prior to that particular period of data collection.

3.7 Data analysis

Collected data was analyzed to determine: 1) cover, 2) plug survival, 3) success of seeded species, and to measure biological diversity using 4) species richness and 5) species evenness.

3.7.1 Cover

The absolute cover of all present species was calculated as a percentage of the total surface area for each platform covered by vegetation for each data collection period. The absolute cover of the 5 control platforms and 5 treatment platforms for each data collection period was averaged and the mean absolute cover (MAC) was compared for each data collection period and tested for significant differences using a type 2 T-test.

Relative cover was defined as the proportion of the absolute cover contributed by a certain species or group of species. Relative cover was used to measure the success of seeded species between treatments by comparing the amount of absolute cover contributed by each species planted by seed. Relative cover was also used to compare the amount of absolute cover contributed by colonizing species with the amount of absolute cover contributed by all planted species between treatments.

3.7.2 Plug Survival

A measure of plug survival was determined by calculating the percentage of individuals from each plug species from each treatment that survived from initial planting on the platform through September 2012. The average final health rating of surviving individuals was calculated by averaging the health scores (see section 3.6) that each of the surviving individuals received during the final data collection period of September. The surviving individuals' final individual health ratings were tested for significant differences between control and treatment using a type 2 T-test which was performed using Microsoft Excel 2010.

3.7.3 Success of Seeded Species

Success of seeded species was defined as germination and survival until dormancy or germination and survival until flowering. The degree of success among seeded species was measured based on the highest mean relative cover (MRC) achieved by each successful species within a treatment. There were some false highest MRC data due to early data collection periods with high numbers of seedlings that did not reach maturity. These percentages were disregarded in favor of later data which more accurately represent successful individuals. The highest mean percent cover of successful seeded species was compared across control and treatment using a type 2 T-test in order to determine if seeded species were more or less successful between control and treatment. The T-tests were performed using Microsoft Excel 2010.

3.7.4 Biological Diversity

The biological diversity of the resultant plant communities was measured using estimates of plant species richness and evenness. Absolute plant species richness, expressed as species density (Magurran, 2003), was determined by recording all species present on each platform and aggregating the totals from each data collection period. Species evenness was measured using Simpson's evenness index

$$E_{1/D} = \frac{(1/D)}{S} \tag{3.1}$$

(Smith and Wilson, 1996) where S is the total number of species and D is Simpson's index of dominance

$$D = \sum p_i^2 \tag{3.2}$$

where pi represents the proportion of the i-th species (Magurran, 2003, Simpson, 1949). A mixed model ANOVA was used to determine whether the effects of treatment, month, and platform influenced the dependent variables of species richness and species evenness, and whether these levels had any interaction between treatments across month. The treatment and month were fixed factors used to examine any differences between treatment groups across month. Since the potential for variability between platforms existed, this factor was nested and randomized to account for any variability across platforms across months within this experiment. In order to determine if these data met the requirements for an ANOVA, a Shapiro-Wilkes goodness of fit test was used to determine if the treatment's species richness and evenness means were normally distributed across month. A Bartlett's test was used in order to determine if the treatment's species richness and evenness means had equal variances across months. A Tukey's HSD post hoc analysis was used to determine whether there was any significant difference among the groups across month. JMP® 10, a statistical analysis software by SAS, was used to perform the mixed-model ANOVA, the Shapiro-Wilkes goodness of fit test, the Bartlett's test and the Tukey's HSD post hoc analysis.

3.7.5 Control and Treatment Media Weight Comparison

Measurements of the control and treatment media weights were performed and the results were compared. Four cups (.95 dm3) of the ERTHHydrocks Lightweight Soil Media-Extensive and 4 cups (.95 dm3) of the treatment mixture containing native prairie soil were placed in containers with drainage holes and weighed when dry and then again after saturation with water. Time was allowed for excess water to drain from the media. This trial was replicated 5 times. The results were averaged and the percentage of weight gained due to water retention was calculated for each treatment. The dry and wet weights of each media per ft^3 (.028 m^3) were calculated.

CHAPTER IV

RESULTS

4.1 Cover

The mean absolute cover (MAC) of all present species across both experimental and control platforms fluctuated between data collection periods. Both control and experimental platforms had achieved 60% or better ATC between October 2011, the date which planting was completed, and March 2012, the date of the first data collection. Both experimental and control platforms experienced an overall decrease in MAC for the data collection periods of May and July before both reaching 90% or better MAC for the September data collection. The MAC for the 5 control platforms was estimated to be 62%, 57%, 39% and 90% for the data collection periods of March, May, July and September, respectively. The MAC for the 5 experimental platforms was estimated to be 60%, 47%, 33% and 93% for the data collection periods of March, May, July and September, respectively.

In testing for statistically significant difference between the MAC across control (n=5) and treatment (n=5) platforms for each data collection period, the α value was set at 0.05. The T-test returned values $t(4) = .187$, $p = .86$, resulting in failure to reject the null hypothesis that there was no significant difference in MAC between control and treatment across the four data collection periods.

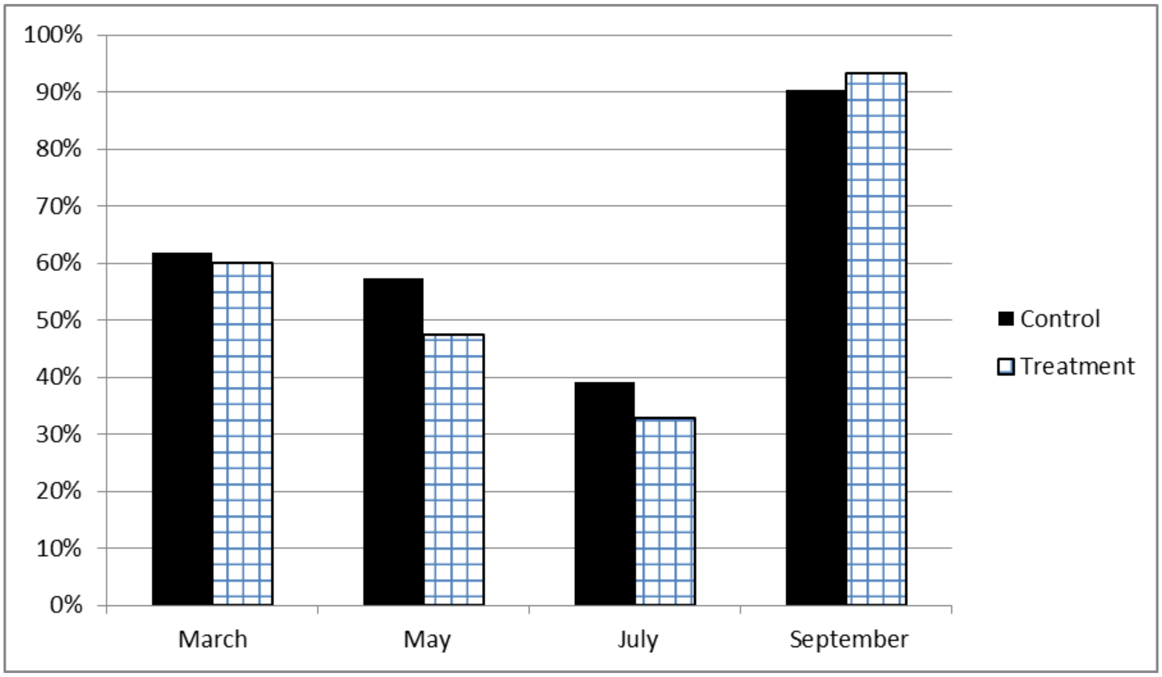

Figure 4.1 Mean Absolute Cover, Given as a Percentage of Total Surface Area, Across Treatment for Each Data Collection Period.

4.2 Plug Survival

Sideoats grama (*Bouteloua curtipendula*) and yellow coneflower (*Ratibida pinnata*) were the only species with individuals that survived through September, 2012. All plugs of blue mistflower (*Conoclinium coelestinum*), purple coneflower (*Echinacea purpurea*) and heath aster (*Symphiotrichum ericoides*) experienced 100% mortality rates across both treatments. Among the 5 plugged species, sideoats grama clearly exhibited the best rates of survival across both treatments. Sideoats grama demonstrated an 82.4% rate of survival on control platforms and a 55.6% rate of survival on the experimental treatment platforms. Surviving individuals of sideoats grama from the control platforms exhibited an average final health rating of 2.1 (on a scale of 0 to 3, see section 3.6) and surviving individuals from the experimental platforms exhibited an average final health rating of 2.4, indicating overall good health across both treatments. In testing for significant difference between the final individual health ratings between control (n=13) and treatment (n=10), the T-test returned values $t(21) = -.899$, $p = .379$, resulting in failure to reject the null hypothesis that there was no significant difference in final health ratings between surviving sideoats grama individuals from the control and treatment platforms. The only other successful plug species, yellow coneflower (*Ratibida pinnata*), experienced much higher mortality. Two out of 20 individuals of yellow coneflower survived on the control platforms, demonstrating a 10% rate of survival, whereas only 1 individual out of 19 survived on the experimental platforms, demonstrating a 5.26% rate of survival. The two surviving individuals on the control platforms exhibited an average health rating of 1.75 and 2.0, respectively. However, the individual with an average health rating of 1.75 received scores of 1 and 2 during the months of March and May, respectively, and then received consistent scores of 3 thereafter. The other individual received consistent scores of 2 for all data collection periods. The lone surviving individual of yellow coneflower from the treatment platforms received consistent scores of 3 for all data collection periods, indicating perfect health. Due to the lack of sufficient data points, no test for significant difference between final health ratings of surviving yellow cone flower individuals was performed.

4.3 Success of Seeded Species

Of the 22 seeded species, 9 species exhibited varying degrees of success. These species were: sideoats grama (*Bouteloua curtipendula*), partridge pea (*Chamaecrista* *fasciculata*), lanceleaf coreopsis (*Coreopsis lanceolata*), tickseed coreopsis (*Coreopsis tinctoria*), yellow sneezeweed (*Helenium amarum*), lemon beebalm (*Monarda citriodora*), evening primrose (*Oenothera speciosa*), black eyed susan (*Rudbeckia hirta*) and little bluestem (*Schizachyrium scoparium*). However, *Bouteloua curtipendula* could not be included in the analysis due to an oversight in data collection. All successful species except *Oenothera speciosa* and *Chamaecrista fasciculata* consistently exhibited a higher mean relative cover (MRC) in the control platforms than the treatment platforms. *Monarda citriodora* achieved the highest MRC of all successful species with 31% in the control platforms and 19% in the treatment platforms (Tbl. 4.1). The t-test for *Monarda citriodora* ($\alpha = 0.05$) returned values $t(8) = 2.559$, $p = 0.034$, resulting in rejecting the null hypothesis that there was no significant difference in success between control and treatment. *Helenium amarum* achieved 7.16% MRC within the control and 2.43% within the treatment. The t-test for *Helenium amarum* ($\alpha = 0.05$) returned values $t(8) = 1.732$, $p = 0.122$, resulting in failure to reject the null hypothesis that there is no significant difference between control and treatment. *Coreopsis lanceolata* achieved 4% MRC within the control and 2% within the treatment. The t-test for *Coreopsis lanceolata* ($\alpha = 0.05$) returned values $t(8) = 1.618$, $p = 0.144$, resulting in failure to reject the null hypothesis that there is no significant difference between control and treatment. *Rudbeckia hirta* achieved 2% MRC within the control and .7% within the treatment. The t-test for *Rudbeckia hirta* ($\alpha = 0.05$) returned values $t(8) = 2.709$, $p = 0.027$, resulting in rejecting the null hypothesis that there is no significant difference between control and treatment. *Schizachyrium scoparium* achieved 2% MRC within the control and .1%

within the treatment. The t-test for *Schizachyrium scoparium* ($\alpha = 0.05$) returned values $t(8) = 2.445$, $p = 0.040$, resulting in rejecting the null hypothesis that there is no significant difference between control and treatment. *Coreopsis tinctoria* achieved .5% MRC within the control and .3% within the treatment. The t-test for *Coreopsis* tinctoria ($\alpha = 0.05$) returned values $t(8) = .7109$, $p = 0.497$, resulting in failure to reject the null hypothesis that there is no significant difference between control and treatment. Both *Oenothera speciosa* and *Chamaecrista fasciculata* achieved approximately 1% MRC within both control and treatment platforms.

Table 4.1 Successful Seeded Species, Their Mean Relative Cover, and T-Test Values

Species	MRC control	n	MRC treatment	n	α	df	t-statistic	p	Null Hypothesis
Bouteloua curtipendula	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Chamaecrista fasciculata	1%	5	1%	5	5	8	0.408955	0.693302	Fail to Reject
Coreopsis lanceolata	4%	5	2%	5	5	8	1.618315	0.144255	Fail to Reject
Coreopsis tinctoria	.5%	5	.3%	5	5	8	0.710932	0.497327	Fail to Reject
Helenium amarum	7.16%	5	2.43%	5	5	8	1.731624	0.121582	Fail to Reject
Monarda citriodora	31%	5	19%	5	5	8	2.558548	0.033723	Reject
Oenothera speciosa	1%	5	1%	5	5	8	-0.55337	0.595133	Fail to Reject
Rudbeckia hirta	2%	5	.7%	5	5	8	2.70928	0.026688	Reject
Schizachyrium scoparium	2%	5	.1%	5	5	8	2.444846	0.040259	Reject

4.4 Colonizing Species

A total of 67 species that were not introduced by direct seeding or plug were recorded across all data collection periods and both treatments. Of those 67 species, 21 were not identified because they existed as small seedlings or rosettes that were short lived and did not reach a level of maturity required for normal identification. They were therefore recorded as unknown species. These species still contributed to estimates of cover and biodiversity, however. In all but one instance, unknown species made very little contribution to cover, as they were represented as one to three seedlings. The one instance where an unidentified species significantly contributed to cover occurred in March on 2 of the treatment platforms where a contiguous area of unknown grass seedlings germinated but died before the next data collection. Interestingly these two areas were later heavily colonized by *Sporobolus* sp. Colonizing species as a whole made up the majority of the total cover for both treatments across all data collection periods except for the control platforms in May.

Of the 46 identified species (Tbl. 4.3), 18 were present only on the treatment platforms and therefore there was a high likelihood that they were introduced in the seed bank of the native prairie soil. Twenty one species were present on both the control and treatment platforms and therefore presumed to have either spontaneously colonized the platforms, or been present in the commercially available media. Seven species were present exclusively on the control platforms (Tbl. 4.2)

Table 4.2 Names of Identified Colonizing Species, Their Location, Native Status and Form.

Species	Common Name	Control	Treatment	Native/ Exotic	grass/ forb
Bromus sp.	brome	*	*	N/A	grass
Cerastium glomeratum	mouse-ear chickweed	*	*	Exotic	forb
Chamaesyce maculata	spotted sand mat	*	*	Native	forb
Chamaesyce nutans	eyebane		*	Native	forb
Chamaesyce sp.	spurge	*	*	N/A	forb
Conyza canadensis	horseweed	*	*	Native	forb
Croton monanthogynus	prairie tea		*	Native	forb
Gamochaeta purpurea	spoonleaf purple everlasting	*		Native	forb
Geranium dissectum	cutleaf geranium	*	*	Exotic	forb
Hordeum pusillum	little barley	*		Native	grass
Lamium amplexicaule	henbit	*	*	Exotic	forb
Leptochloa panicea	mucronate sprangletop		*	Native	grass
Lespedeza cuneata	sericea lespedeza	*	*	Exotic	forb
Medicago lupulina	black medick		*	Exotic	forb
Mollugo verticillata	green carpetweed	*	*	Native	forb
Nuttallanthus texanus	Texas toadflax	*	*	Native	forb
Oenothera laciniata	cutleaf evening primrose	*		Native	forb
Oxalis corniculata	creeping woodsorrel	*	*	Native	forb
Oxalis corniculata var. atropurpurea	creeping woodsorrel		*	Native	forb
Panicum dichotomiflorum	Fall panicgrass		*	Native	grass
Paspalum sp.	paspalum	*	*	N/A	grass
Plantago lanceolata	narrowleaf plantain	*		Exotic	forb
Poa annua	annual bluegrass	*	*	Exotic	grass
Ranunculus fascicularis	early buttercup	*	*	Native	forb
Rumex sp.	dock		*	N/A	forb
Solidago sp.	goldenrod		*	N/A	forb
Sporobolus sp.	dropseed		*	Native	grass
Steinchisma hians	gaping grass	*		Native	grass
Stellaria media	common chickweed	*	*	Exotic	forb
Taraxacum sp.	dandelion	*	*	N/A	forb
Triodanis perfoliata	venus' looking glass	*	*	Native	forb
Trifolium repens	white clover	*	*	Exotic	forb

Table 4.2 (continued)

Species	**Common Name**	**Control**	**Treatment**	**Native/ Exotic**	**grass/ forb**
Trifolium resupinatum	persian clover		*	Exotic	forb
Ulmus sp.	Elm	*		N/A	tree
Verbena simplex	narrowleaf vervain		*	Native	forb
Vicia vilosa	winter vetch		*	Exotic	forb
Vulpia octoflora	six weeks fescue	*	*	Native	grass

Table 4.3 Proportions of Colonizing Species According to Location and Native Status

	Total	Native Status Unknown	Native	% Native	Exotic	% Exotic
Species Not Introduced by Direct Seeding or Plug	46	7	24	62%	15	38%
Species Present in Control and Treatment	21	4	9	53%	8	47%
Species Present in Treatment Only	18	2	10	63%	6	38%
Species Present in Control Only	7	1	5	83%	1	17%

Of the 21 identified colonizing species that were present on both control and treatment platforms, 4 species were only identified to genus level and therefore were unable to be determined to be native or exotic, 9 species were determined to be native and 8 were determined to be exotic. Of the 18 identified colonizing species present only on the treatment platforms, 2 were only identified to genus level, 10 were determined to be native and 6 were determined to be exotic. Of the 7 identified colonizing species present only on the control platforms, 1 was only identified to the genus level, 5 were determined to be native and 1 was determined to be exotic. The lone species from the control platforms that was only identified to genus level was an elm tree (Ulmus sp.) seedling that germinated from the potting soil of a plug before placement on the platform.

Colonizing species made significant contributions to the absolute cover on all platforms across both treatments and for all data collection periods. For the data collection periods of March, July and September, colonizing species exhibited at least 73.9% MRC on the control platforms. For the data collection period of May, colonizing species saw a drastic drop in coverage, exhibiting only 19.5% MRC on the control platforms (Fig. 4.2). For the data collection periods of March, July and September, colonizing species exhibited at least 79.9% MRC on the treatment platforms. Colonizing species also saw a drop in coverage on the treatment platforms for the data collection period of May, exhibiting only 52.4% MRC (Fig. 4.3).

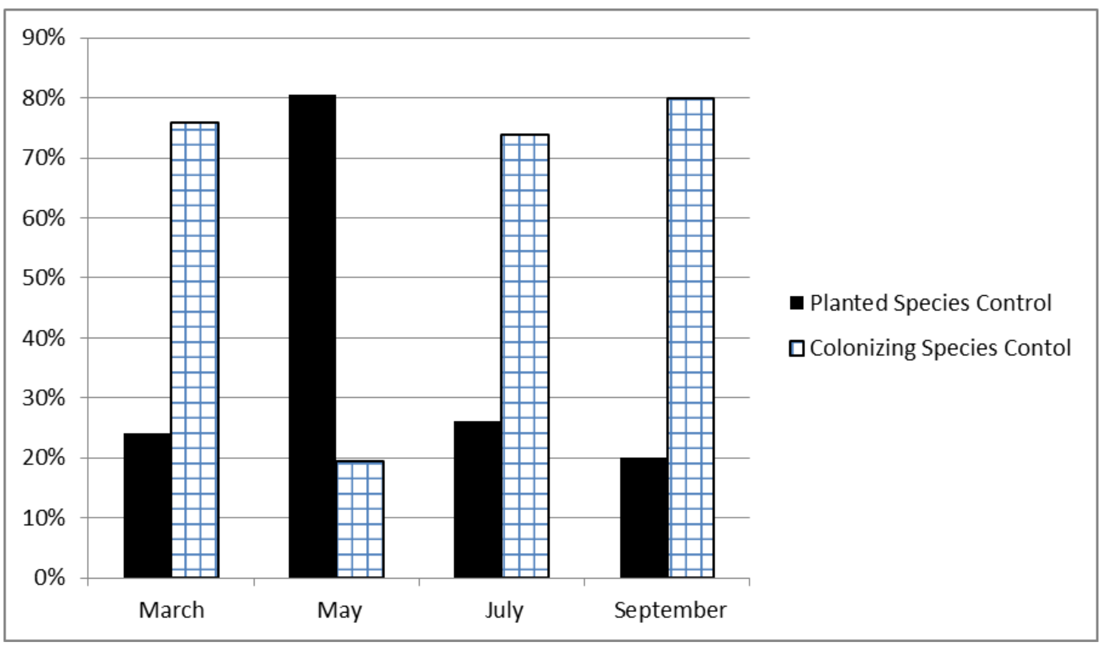

Figure 4.2 Mean Relative Cover of Colonizing vs. Planted Species Across Control Platforms.

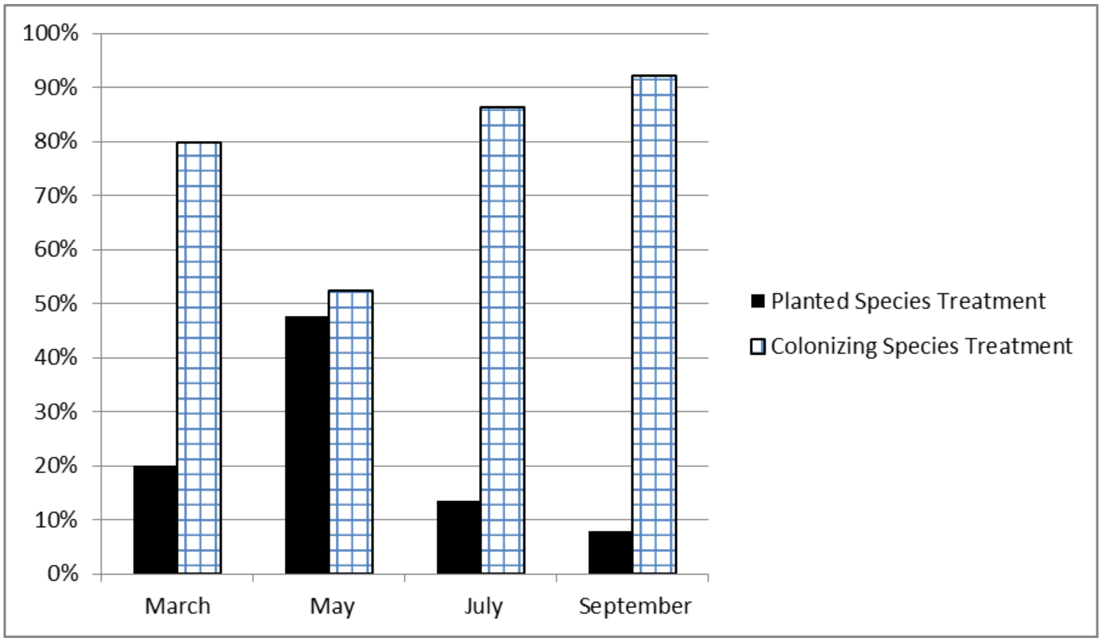

Figure 4.3 Mean Relative Cover of Colonizing vs. Planted Species Across Treatment Platforms.

Most of the colonizing species made only minor contributions to the total cover because they existed as only a few representatives on various platforms. However, some species experienced seasonal flourishes across both treatments during certain data collection periods, vigorously filling available space and completing their life cycle before disappearing. For example, two species of chickweed, *Cerastium glomeratum* and *Stellaria media*, on average accounted for 50.7% of the total cover on the control platforms and 31.4% of the total cover on the treatment platforms in March. Also in March, two species of clover, *Trifolium repens* and *Trifolium resupinatum*, experienced a lesser flourish and accounted for 5.3% of the total cover on the control platforms and 6.0% of the total cover on the treatment platforms. In July, a native species of crabgrass, *Digitaria ciliaris*, on average accounted for 51.6% of the total cover on the control platforms and 22.6% of the total cover on the treatment platforms. In September, two similar species of spurge, *Chamaesyce maculata* and *Chamaesyce* sp., together accounted for 17.6% of the total cover on the control platforms and 10.6% of the total cover on the treatment platforms. Another colonizing species, black medick (*Medicago lupilina*) accounted for 10.8%, and 11.6% of the total cover on the treatment platforms in March and May, respectively. Black medick was only nominally present on one control platform. Two species of grass which were present only on the treatment platforms, *Festuca rubra* and *Sporobolus* sp., aggressively colonized one platform each but were also present on others.

4.5 Species Richness

The α (alpha) value was set at 0.05 for the statistical analyses used. The Shapiro-Wilkes goodness of fit test returned a p-value of .09 for the control (n=20) and a p-value of .22 for the treatment, resulting in no significant departure from normality and therefore acceptance of the null hypothesis that the species richness means were normally distributed across month. The mean richness value for the control platforms was 15.25, with a standard deviation of 5.53. The mean richness value for the treatment platforms was 18.1, with a standard deviation of 6.54. The Bartlett's test returned a p-value of $p = 0.47$ signifying that there was no significant difference in variance between the mean richness values of the control and treatment groups. The overall ANOVA model for richness returned values $R2 = 0.93$, $F(15, 24) = 21.58$, $p < 0.0001$, revealing that there is a significant difference between the factors of treatment and month in relation to species richness, leading to the rejection of the null hypothesis that there was no significant difference. In testing the null hypothesis that there was no significant interaction between treatment and month, the ANOVA revealed a p-value of 0.22 ($F(3, 24) = 1.58$, $p = 0.22$), resulting in retaining the null hypothesis. The treatment group had overall higher richness values than did the control group across all months. However, for the summer months of May and July the richness values for control and treatment were very similar (Fig. 4.2). Since the two plotted lines of richness values do not intersect, no significant interaction between the factors of treatment and month was indicated.

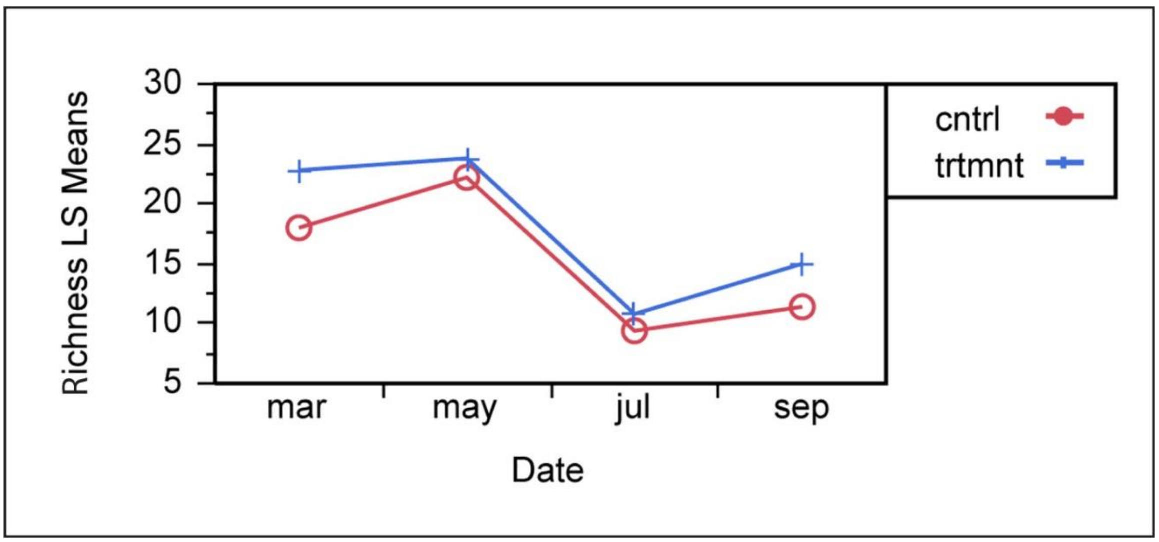

Figure 4.4 Mean Least Square Species Richness Values for Control and Treatment Roofs by Month

In testing the null hypothesis that there was no significant difference between control and treatment groups in terms of least square (LS) mean richness, the ANOVA returned a p-value of 0.09 ($F(1, 24) = 3.61$, $p = 0.09$), indicating that there is no significant difference between control and treatment LS mean richness values, which lead to accepting the null hypothesis. In testing the null hypothesis that there was no significant difference across months in terms of LS mean richness, the ANOVA returned a p-value of less than 0.0001, ($F(3, 24) = 85.81$, $p < 0.0001$), indicating that there is a highly significant difference across month in relation to LS mean richness values, which lead to rejecting the null hypothesis. A Tukey's HSD post hoc analysis revealed that there was a significant difference in relation to species richness between every date. May had the highest significant overall LS mean richness value, followed by March, then September and July, in descending order.

4.6 Species Evenness

The α value was set at 0.05 for the statistical analyses used. The Shapiro-Wilkes goodness of fit test returned a p-value of 0.06 for the control (n=20) and a p-value of 0.08 for the treatment, resulting in no significant departure from normality and therefore a failure to reject the null hypothesis that the mean evenness values are normally distributed across month. The mean evenness value for the control group was 0.32, with a standard deviation of 0.11. The mean evenness value for the treatment platforms was 0.36, with a standard deviation of 0.12. The Bartlett's test returned a p-value of $p = 0.09$ signifying that there was no significant difference in variance between the mean evenness values of the control and treatment groups. The overall ANOVA model for evenness returned values $R2 = 0.665$, $F(15, 24) = 3.17$, $p = 0.006$ revealing that there is a significant difference between the factors of treatment and month in relation to evenness, leading to the rejection of the null hypothesis that there was no significant difference. In testing the null hypothesis that there was no significant interaction between treatment and month in relation to evenness, the ANOVA revealed a p-value of 0.36 ($F(3, 24) = 1.13$, $p = 0.36$), resulting in accepting the null hypothesis. The treatment group showed higher evenness values than the control group for March, May and July. In the month of September however, the control group expressed a higher LS mean evenness value indicating that there was an interaction between the factors of treatment and month in September (Fig 4.3).

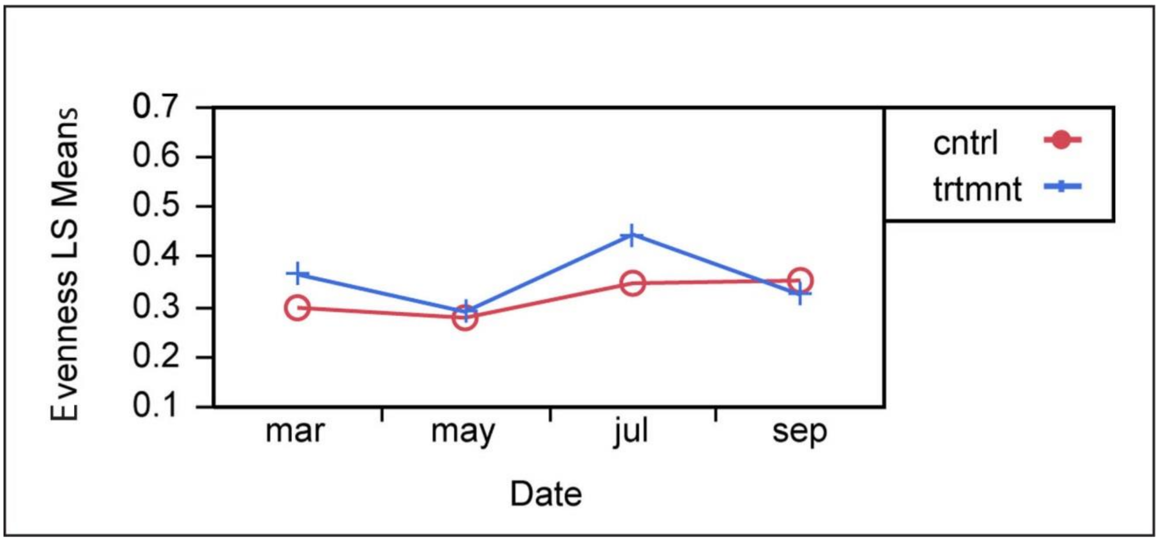

Figure 4.5 Mean Least Square Evenness Values for Control and Treatment Roofs by Month.

In testing the null hypothesis that there was no significant difference between control and treatment groups in terms of LS mean evenness, the ANOVA returned a p-value of 0.51 ($F(1, 24) = 0.47$, $p =0.51$), indicating that there is no significant difference between control and treatment LS mean evenness values, which lead to retaining the null hypothesis. In testing the null hypothesis that there was no difference across months in terms of LS mean evenness, the ANOVA returned a p-value of 0.05 ($F(3, 24) = 3.01$, $p = 0.05$), indicating that there is a significant difference across month in relation to LS mean evenness values, which lead to rejecting the null hypothesis.

4.7 Visual Characteristics

The platforms saw a flourish of cover provided by a suite of early season (C3) species in late winter through mid-spring. This period was marked by lush green foliage and a variety of flowering forbs. This was followed by a transitional period beginning in May in which the platforms experienced a decline in cover and had an overall arid appearance. In mid to late summer, the platforms experienced a second flourish of cover provided by a suite of late season (C4) species, this time mostly grasses. By September, the second flourish of verdant growth had matured into a drier aesthetic with attractive grass seedheads and yellow sneezeweed blooms providing a familiar autumn meadow pattern. By late autumn this second flourish gave way to a shaggy, browned-out appearance, with the dormant foliage and seedheads of sideoats grama and little bluestem providing the only visible structure across the platforms. By February of 2013, the thatch from the previous year had subsided and the platforms were nearly completely covered by small green seedlings and rosettes. It appears that winter is a verdant period for green roofs in humid subtropical climates and that many cool-season plants can complete their life cycles on them before the onset of hot weather. It should be noted that many forbs that were present in the first year but didn't bloom, bloomed profusely in the second year of the experimental green roofs used in this study.

4.8 Media Weights

On average, the control media weighed 49.67 lbs/ft^3 (795.64 kg/m^3) when dry and 59.54 lbs/ft^3 (953.74 kg/m^3) when saturated with water (Tbl. 4.4). The treatment media weighed 55.36 lbs/ft^3 (886.77 kg/m^3) when dry and 73.31 lbs/ft^3 (1174.31 kg/m^3) when saturated. The treatment media weighed 11.5% more than the control media when dry and 23% more than the control media when wet. From this it was determined that the treatment media retained 12.5% more water than the control media.

Table 4.4 Wet and Dry Weight Comparison of Control and Treatment Media

	control		**treatment**	
	dry	**wet**	**dry**	**wet**
trial 1	1.69 lbs.	2.06 lbs.	1.75 lbs.	2.28 lbs.
trial 2	1.81 lbs.	2.13 lbs.	2 lbs.	2.75 lbs.
trial 3	1.63 lbs.	2.06 lbs.	1.94 lbs.	2.69 lbs.
trial 4	1.5 lbs.	1.78 lbs.	1.75 lbs.	2.13 lbs.
trial 5	1.69 lbs.	1.94 lbs.	1.81 lbs.	2.38 lbs.
average	**1.66** lbs.	**1.99** lbs.	**1.85** lbs.	**2.45** lbs.
s. deviation	.1 lbs.	.12 lbs.	.1 lbs.	.24 lbs.
% difference			11.5% heavier when dry	23% heavier when wet
wet difference		.33 lbs.		.6 lbs.
% gain		19.9% weight gained when wet		32.4% weight gained when wet
% difference			12.5% more water held	
wt. per ft.^{3}	49.67 lbs.	59.54 lbs.	55.36 lbs.	73.31 lbs.

4.9 Climatic Conditions

Table 4.4 provides the monthly climatic conditions for Starkville, Mississippi for January, 2013 until the end of data collection in September of that year.

Table 4.5 Monthly Average Maximum and Minimum Temperatures in °F, Monthly Precipitation in inches, and Most Consecutive Days with Zero Precipitation, January – September, 2013

	Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**
Avg. Max	62.2	59.8	76.2	77.6	85.8	89.0	92.9	88.9	85.6
Avg. Min	36.0	38.6	53.5	54.6	63.8	67.2	72.5	69	61.8
Precipitation	3.01	4.05	7.39	3.74	3.30	2.84	9.34	7.76	5.36
Dry Days	7	5	6	12	8	22*	22*	6	10

(National Oceanic and Atmospheric Administration, 2014)
* - Denotes that the dry period overlapped two different months.

CHAPTER V

DISCUSSION

5.1 Cover

In March 2012, 4 months after planting, the plug species were only beginning to revegetate and, other than sideoats grama, which retained much of its above ground mass through winter, contributed very little to absolute cover. Yet all platforms had achieved between 50% and 68% absolute cover by this time due to the cover provided by colonizing species. Over the next two data collection periods, May and July, all platforms saw a gradual decrease in absolute cover. By September, however, all platforms had experienced a drastic escalation in absolute cover, with all platforms achieving between 81% and 96% absolute cover. This indicates that two vegetative flourishes occurred, one in late winter/early spring, and another in late summer/early fall. In contrast, a related study conducted at the same site analyzed the rate of establishment of 4 different sedum species (without colonizing vegetation) on green roofs with 2% and 33% slopes and 4 in (10.6 cm) and 6 in (15.24 cm) soil media depths and found that with occasional irrigation, a maximum amount of cover of 56.56% was established on the 6 in media depth at 2% slope in the second year of establishment (Kordon, 2012). For comparison, Dunnett's 2001-2006 study conducted in Sheffield, England compared the cover provided by a mixture of succulent, non-succulent and spontaneously colonizing vegetation at 100 mm

(3.94 in) and 200 mm (7.87 in) soil media depths and found that the deeper substrate achieved 85% cover and the shallower substrate achieved 58% cover during the first year (Dunnett and Allison, 2004). Monterusso et al. (2005) observed 100% coverage by *Sedum* spp. by the third year on green roofs in Michigan with the aid of regular irrigation through a 2-year establishment period.

5.2 Plug Survival

Sideoats grama (*Bouteloua curtipendula*) planted as plugs, clearly proved to be successful on both control and treatment platforms. There was no statistically significant difference in the survival or final average health ratings of sideoats grama between control and treatment platforms, and all surviving individuals expressed reasonably good health. Also, all surviving individuals of sideoats grama bloomed and set seed on the platforms.

The only other surviving plugged species, yellow coneflower (*Ratibida pinnata*), experienced much higher mortality rates than sideoats grama. Interestingly, all 3 surviving yellow coneflower individuals exhibited good health throughout the experiment. The one surviving individual from the treatment platforms exhibited perfect growth and health for all data collection periods, even in May and July when most other species were experiencing stress from hot and dry weather. This adds to the findings of Monterusso et al. (2005) who observed total mortality in all non-succulent species planted as plugs except *Tradescantia ohiensis* and *Coreopsis lanceolata*. However, Monterusso et al. found that the surviving individuals of these two species experienced a drastic decline in health after an initial establishment period of regular irrigation.

Even among perfectly healthy examples, surviving individuals of both sideoats grama and yellow coneflower exhibited an overall dwarfed growth habit compared to the habits of the same species superficially observed growing at ground level. This is likely a response to dry conditions, low fertility or both.

5.3 Success of Seeded Species

Monterusso et al. (2005) reported lanceleaf coreopsis (*Coreopsis lanceolata*) and Ohio spiderwort (*Tradescantia ohiensis*) successfully establishing on green roofs by seed in Michigan. The results of the experiment in this thesis confirm that seeding can be an effective means of establishing lanceleaf coreopsis and other native, non-succulent herbaceous species on unirrigated green roofs for a humid subtropical climate. In this instance, nine of the 22 species planted by seed experienced at least some success. However, these 9 species can be further subdivided into 2 groups: those that were clearly successful and those that were marginally successful. As previously stated, success of a seeded species was defined as germination and survival until flowering, or germination and survival until dormancy. The marginally successful species achieved germination and survival until flowering, but exhibited low rates of establishment and deficient growth in the first year. There were 4 marginally successful species: partridge pea (*Chamaecrista fasciculata*), tickseed coreopsis (*Coreopsis tinctoria*), evening primrose (*Oenothera speciosa*), and black-eyed susan (*Rudbeckia hirta*). One other marginally successful species, Illinois bundleflower (*Desmanthus illinoiensis*), did germinate and persist but did not flower in the first year.

The five seeded species that were clearly successful included sideoats grama (*Bouteloua curtipendula*), yellow sneezeweed (*Helenium amarum*), lemon beebalm (*Monarda citriodora*), little bluestem (*Schizachyrium scoparium*), and lanceleaf coreopsis (*Coreopsis lanceolata*). Even though sideoats grama germinated and survived, it could not be included in the statistical analysis due to an oversight in data collection. Recruitment of new clumps of sideoats grama was evident but no differentiation was made between the cover provided by individuals planted as plugs and new clumps when they occurred in the same grid cell. Furthermore, it was impossible to determine if new clumps of sideoats grama resulted from the hand sown seeds or were the offspring of the individuals set out as plugs, as many had set seed prior to planting. Yellow sneezeweed, an annual (USDA, NRCS, 2013), bloomed profusely in late summer and early fall and set seed. Lemon beebalm, which can have an annual or perennial duration (USDA, NRCS, 2013), bloomed profusely in early summer and set seed but either died or entered early dormancy by mid-June. Several small clumps of little bluestem were recorded across control and treatment platforms and one clump, on a control platform, bloomed.

All successful seeded species exhibited higher mean relative cover (MRC) on the control platforms than on the treatment platforms except evening primrose and partridge pea, which exhibited equal MRC between treatments. Dunnett (2006) suggested that incorporating mycorrhizal fungi and other micro-fauna through the addition of native soil may be beneficial to the establishment and growth of vegetation on green roofs. These findings seem to imply that the opposite is true however there are many other variables that may have arisen from soil modification. Sutton (2008) found that the interaction between native soil additive and a hydro-absorbent polymer gel, which serves to retain moisture, did enhance the vigor of native grasses, sedges and forbs. This suggests that moisture availability may have limited the effectiveness of the micro-flora present in the native soil additive in this capacity. The most meaningful implication of this result is that adding native soil to green roofs will not necessarily have a beneficial effect on plant survival or establishment.

5.4 Colonizing Species

The results of this study confirm Kadas' (2006) suggestion that plants will move in and quickly establish communities in available niches on green roofs for a humid subtropical climate. Altogether, 46 colonizing species were identified on the platforms in the first year of study. Of those 46, 18 were very likely introduced in the native soil seed bank, leaving 28 species that likely colonized the roofs from extraneous sources. This can be compared to Dunnett et al.'s (2008) results of 35 colonizing species recorded over 6 growing seasons. Some of the same plant genera that colonized Dunnett's green roofs in England also colonized the roofs in this experiment and included: *Medicago*, *Taraxacum*, *Trifolium*, *Poa*, *Rumex*, *Gallium*, *Plantago* and *Vicia*. Dunnet et al. (2008) predicted that spontaneously colonized green roofs would be temporally dynamic in nature, particularly in their establishment phase and that ruderal species would be the main beneficiaries of local seed bank sampling (Dunnett, 2006). Both of these phenomena were observed in this experiment. Dunnett et al. (2008) also stated that by definition, ruderal species are "weedy" and have little aesthetic potential. Counter to this finding, a number of colonizing species were noted for potential value for use on green roofs because of consistent health during hot and dry weather and aesthetic potential such as exceptional blooms, seeds or texture (Tbl. 5.1, Tbl. 5.2).

In general, grasses excelled on both control and treatment platforms. All of the colonizing grass species that survived long enough to be identified were noted for being quite hardy in that they never seemed to struggle through the hottest and driest periods other than normal signs of heat stress. Although many of the colonizing grass species from this study are perhaps aesthetically nondescript, some exhibited striking foliage and texture such as purple love grass (*Eragrostis spectabilis*), a species commonly sold as an ornamental landscape plant, and fall panic grass (*Panicum dichotomiflorum*). Although there is a distinct lack of existing research regarding non-succulent plant performance on unirrigated green roofs, Dvorak and Volder (2013) also experienced success with a species of native grass, *Nassella tenuissima*, on unirrigated green roofs in a dry subtropical climate.

Table 5.1 Notable Characteristics, Location, Period of Visibility and Native Status of Colonizing Grasses with Potential for Green Roof Applications

Common Name	Scientific Name	Health	Flowers/ Seeds	Texture	Control	Treatment	Season (W, SP, S, F)	Native Exotic
purple love grass	*Eragrostis spectabilis*	*	*	*		*	S, F	N
Red Fescue	*Festuca rubra*	*		*		*	S	N
little barley	*Hordeum pusillum*	*	*	*	*		SP	N
mucronate sprangletop	*Leptochloa panicea*	*	*	*		*	F	N
fall panic grass	*Panicum dichotomiflorum*	*		*		*	F	N
Sporobolus sp.	*Dropseed*	*		*		*	S, F	N

Season: W = winter, SP = spring, S = summer, F = fall

Many of the colonizing forbs exhibited either suboptimal or oscillating health throughout their lifespan. Some maintained just enough vigor to flower and complete their life cycles, such as Queen Anne's lace (*Daucus carota*). However some forbs maintained perfect health throughout their period of visibility on the platforms, never showing signs of heat or drought stress. For example, a single individual of narrowleaf vervain (*Verbena simplex*), a local prairie native recruited on a single treatment platform produced supple foliage and striking blooms during a transitional period in May when almost all other species seemed to suffer. Another local prairie native, prairie tea (*Croton monanthogynus*), germinated in May, bloomed, set seed and persisted through fall, never showing signs of stress. Many cool-season grasses and forbs such as texas toadflax (*Nuttallanthus texanus*), venus' looking glass (*Triodanis perfoliata*) and early buttercup (*Ranunculus fascicularis*), experienced good health throughout their life cycles. From this observation, it seems likely that winter and early spring are optimal growing periods on green roofs in humid subtropical climates. Interestingly though, many of these same cool-season species were observed persisting much longer into summer on nearby green roof platforms where they were growing through patches of Sedum cover.

Table 5.2 Notable Characteristics, Location, Period of Visibility and Native Status of Colonizing Forbs with Potential for Green Roof Applications.

Common Name	Scientific Name	Health	Flowers	Texture	Control	Treatment	Season (W, SP, S, F)	Native Exotic
prairie tea	*Croton monanthogynus*	*		*		*	S,F	Native
Queen Anne's lace	*Daucus carota*		*		*	*	SP	Exotic
Texas toadflax	*Nuttallanthus texanus*	*	*		*	*	SP	Native
cutleaf evening primrose	*Oenothera laciniata*		*	*	*		SP	Native
early buttercup	*Ranunculus fascicularis*		*		*	*	W, SP	Native
venus' looking glass	*Triodanis perfoliata*		*		*	*	SP	Native
white clover	*Trifolium repens*		*	*	*	*	W, SP	Exotic
persian clover	*Trifolium resupinatum*		*	*	*	*	W, SP	Exotic
narrowleaf vervain	*Verbena simplex*	*	*	*		*	S	Native

Season: W = winter, SP = spring, S = summer, F = fall

Another group of colonizing species is worth noting due to their tendency to ubiquitously colonize disturbed sites and therefore have a high likelihood of spontaneously colonizing newly constructed green roofs (Tbl. 5.3). Some of these species have aesthetic appeal such as the 2 species of chickweed (*Cerastium glomeratum* and *Stellaria media*) which quickly filled in vacant space in spring and provided a fine-textured ground cover. Another such example is creeping wood sorrel (*Oxalis corniculata*) which germinated and provided small yellow flowers in early spring, disappearing afterwards only to reemerge and bloom again throughout the growing season whenever there was a brief period of damp weather. Some of these ubiquitous species could likely be perceived as weedy or undesirable such as southern crabgrass

(*Digitaria cilliaris*), Paspalum (*Paspalum* sp.) and barnyard grass (*Echinochloa crus-galli*). During the review of green roof literature conducted for this study, no examples of research regarding aesthetic perceptions of colonizing species or spontaneous plant communities were discovered. Dunnett et al. (2008) used physical dimensions (height and spread) and ratio of flowering to non-flowering shoots to measure the aesthetic value of the 15 planted species in their experiment but did not examine the colonizing species in this regard.

Table 5.3 Notable Characteristics, Location, Period of Visibility and Native Status of Ubiquitous Colonizers of Disturbed Sites Present on Platforms

Common Name	Scientific Name	Health	Flowers/ Seeds	Texture	Control	Treatment	Season (W, SP, S, F)	Native Exotic
mouse-ear chickweed	*Cerastium glomeratum*			*	*	*	SP	Exotic
spotted sand mat	*Chamaesyce maculata*				*	*	S, F	Native
southern crabgrass	*Digitaria cilliaris*				*	*	S	Native
barnyard grass	*Echinochloa crus-galli*	*			*	*	S, F	Exotic
spoonleaf purple everlasting	*Gamochaeta purpurea*			*	*		SP	Native
cutleaf geranium	*Geranium dissectum*				*	*	SP	Exotic
henbit	*Lamium amplexicaule*		*		*	*	W	Exotic
green carpetweed	*Mollugo verticillata*				*	*	S	Native
creeping woodsorrel	*Oxalis corniculata*	*	*		*	*	Sp, S, F	Native
paspalum	*Paspalum sp.*				*	*	S, F	N/E
narrowleaf plantain	*Plantago lanceolata*		*		*		SP, S	Exotic
annual bluegrass	*Poa annua*				*	*	W	Exotic
common chickweed	*Stellaria media*			*	*	*	SP	Exotic
dandelion	*Taraxacum sp.*		*		*	*	W, SP	N/E
white clover	*Trifolium repens*		*		*	*	W, SP	Exotic
winter vetch	*Vicia vilosa*		*		*	*	SP	Exotic

Season: W = winter, SP = spring, S = summer, F = fall

5.5 Biodiversity

The experiment described by Dunnett and Alison (2004) and Dunnett et al. (2008) used the Shannon-Weiner index to measure the biodiversity of a planted and a spontaneously colonizing plant community but did not relate the actual Shannon-Weiner values. Dunnett et al. (2008) did state that even though colonizing plant communities of shallower soil depths scored higher Shannon-Weiner values, the communities had low evenness. Simpson's index and the Shannon-Weiner index are two ways to measure biological diversity commonly used within the ecological literature (Smith and Wilson, 1996). They are heterogeneity measures, single statistics that include information on the richness and evenness components of diversity (Magurran, 2003). Magurran (2003) recommends Simpson's index over the Shannon-Weiner index because it provides a good estimate of diversity at small sample sizes and will rank assemblages consistently (Magurran, 2003). This study reports and compares species richness and evenness separately, and implements a measurement of evenness derived from Simpson's index of diversity, known as $E_{1/D}$ (Smith and Wilson, 1996) or Simpson's evenness (Magurran, 2003). $E_{1/D}$ has an advantage over other evenness measurements in that its value can reach a minimum of 0 and a maximum of 1 (Smith and Wilson, 1996). This means the abundance of all species together is designated as 1.0 or 100% and the relative abundance of each species is given as a percentage of the total (Magurran, 2003). By multiplying $E_{1/D}$ by the total number of species, Simpson's evenness can be converted into Simpson's index of diversity (Smith and Wilson, 1996). Both Simpson's index and Simpson's evenness provide intuitively understandable values that increase as diversity and evenness increase (Magurran, 2003).

Even though the richness of non-plant species was not quantified in this study, similarities to other studies were superficially observed. In their 2004 survey of fauna occurring on the Ford Motor Company green roof in Dearborn, Michigan, built in 2002 Coffman and Davis (2005) found 29 families of winged insects, 7 spider species and 2 species of birds present. A similar study identified fauna on established green roofs in Switzerland and found a rich diversity of vertebrate and invertebrate species using the roofs including 78 spider and 254 beetle species, as well as endangered species of birds (Brenneisen, 2003). During the present study, fungi and non-vascular plants were observed growing on the platforms, as there was at least one species of mushroom and one species of moss noted. Several species of animals were also observed using the platforms. Insects, including pollinators (butterflies, bees and wasps), predators (wasps, dragonflies and praying mantises) and ants were present. A suite of spiders were also observed using the platforms including jumping spiders, orb weavers and beautifully colored crab spiders. Several avian species including the ground-foraging mourning dove were observed. Surprisingly, there was a single species of gastropod found using the roofs. Small snails that were also present on the ground were observed on the roofs throughout the growing season and appeared to be dwelling there.

5.6 Implications for Designers

Unirrigated, low-maintenance green roofs, such as those created in this experiment, have several advantages over those that implement irrigation and frequent weeding. The most immediate advantage is being able to avoid the initial cost of constructing an irrigation system and lower long-term maintenance costs. Depending on project-specific needs, it is possible that such roofs could require no regular plant maintenance whatsoever. A longer term advantage is the potential for achieving high levels of biodiversity through the allowance of spontaneous colonization. Biodiversity is an important aspect of the long-term sustainability of green roofs and needs to be a design goal if a particular green roof is intended to be a low-input, self-sustaining system. Diverse plant communities, In addition to their potential for enhancing green roof services such as stormwater retention, temperature regulation and absorption of solar radiation (Dunnett et al., 2008; McIvor and Lundholm, 2011), have inherent habitat value and perform natural nutrient cycling that replaces the need for supplemental fertilizers (Sutton, 2008). Questions about the long-term sustainability of green roof projects arise also from the use of irrigation (Koehler, 2009), especially potable water. Yet most American green roof projects represented in the literature, even those promoted as recreations of biodiverse habitats such as prairies, seem to adhere to a rigid protocol of irrigation and removal of colonizing species, viewing them as weeds (e.g. Lindell, 2008; Armstrong, 2009; Griswold, 2010). Thus designers and researchers seem to be prioritizing aesthetic goals in ways that limit biodiversity and the self-sufficiency of green roofs. The following sections attempt to explore these conflicts and provide ideas regarding the design of unirrigated, biodynamic green roofs.

5.6.1 Aesthetics

The plant communities observed in this study oscillated between lush green growth, flowering, and drought-induced dieback. Aesthetically speaking, they experienced periods of a dry and arid appearance, periods of verdant green, and at times, beautiful textures and flowering. There was a prolonged period in late fall, before the subsequent wave of verdant growth in winter, when the green roofs were completely brown and appeared dead. The perceptual conflict between aesthetics and natural ecological processes of vigor and decline, and natural succession could be a barrier to the implementation of such biodynamic roofs. Mozingo (1997) recognized this conflict and argued that the success of ecological design depends on the abilities of designers to produce beautiful projects that resonate with the public. A perceived challenge of designing biodynamic green roofs is that one cannot expect a static and controlled aesthetic expression from plants like that often achieved in ground level landscaping through intensive maintenance. This is not a disadvantage but rather a design challenge. Furthermore, these concerns are not applicable to many potential green roof projects.

Most green roofs presented in the literature are intended to serve as high-profile pioneer projects with a goal of winning public acceptance. For these projects, it is understandable that maintaining a constant and lush aesthetic be given priority. But there are added design, maintenance, and environmental costs associated with this approach. In order for the aggregated benefits of green roofs to be realized, green roofs must be implemented on a citywide scale with the greatest benefits being reaped from the greening of commercial and industrial rooftops (Carter and Jackson, 2006). This could be achieved through tax incentives or direct legislation for on-site stormwater management (Carter and Fowler, 2008), in which case most property owners would be interested in the lowest possible cost of implementation. A roof-greening scenario like this would include mostly low-profile green roof projects such as big box stores and strip malls, many of which would never even be visible to the public. In these instances, aesthetics would be irrelevant and the results of this study suggest that simply installing a layer of soil media without planting or irrigation could achieve near 100% vegetative cover within the first year. This would provide the enhanced benefits that plant cover contributes to green roof functions (Dunnett, Nagase, Booth and Grime, 2008; Lundholm et al., 2010) and also opportunities for biodiversity.

Another example in which aesthetic primacy would be minimized is when a biodynamic green roof is viewed from far away. In this case, the green roof vegetation would appear as a uniform texture with periodic floral displays. Even during periods of dieback, this would certainly be more interesting than an unchanging view of conventional, tar-covered flat roofs. Biodynamic green roofs do not have to be relegated to unseen or distant rooftops, however. In these cases designers must meet the design challenge of acknowledging the ebb and flow of natural systems.

Joan Nassauer argued that the way to communicate the ecological function of the landscape is by setting expected characteristics of landscape beauty side by side with characteristics of ecological health (Nassauer, 1992). Floral displays and use of color in green roof designs could prove invaluable in this regard (Dunnett, 2006). This could be taken a step further by adapting Bates et al.'s (2009) recommendations for improving biodiversity on green roofs by using topographical variations of soil media depths and cover objects such as wood and stone to produce micro-habitats. For example, cover objects could be designed as sculptural elements that provide Nassauer's "cues to care" (1995) during periods of decline and dormancy. Varying soil media depths will result in correspondingly different plant communities. This could be done in an artful manner to create readily legible patterns that interplay with the abiotic elements in changing ways.

Green roofs can be a true interface between ecological systems and the built environment if designed as such. Like all ecosystems, those on green roofs are fluid and may take a long time to stabilize (Dunnett et al., 2008). In time, unirrigated green roofs can become reservoirs of biodiversity (Brenneisen, 2006) but in order to communicate the beauty of ecological processes, designers must relinquish the approach of absolute control and design for the allowance of change.

5.6.2 Maintenance

Two practical realities of green roof maintenance became clear during this study. One is that without continual and frequent weeding, spontaneous colonization is going to occur. Colonization is in fact necessary in order to achieve levels of plant diversity beyond the planting list. Secondly, irrigation is not necessary for the survival of non-succulent herbaceous plants on green roofs if adequately adapted species are used. There seems to be a general presumption of the need for meticulous maintenance specifications for green roofs (i.e. Lindell, 2008; Griswold, 2010). Some designers seem to be focused on maintaining a static replication of ground level landscaping on green roofs (Butler et al., 2012) at the expense of natural succession and self-sufficiency. For example: in maintaining a green roof designed to be a prairie, Dewey, Johnson and Kjelgren (2004) viewed species that were well enough adapted to the conditions on the roof that they spread from their predetermined positions as being too competitive. An article from *Sustainable Facility*, identified proper green roof maintenance stating that all green roof projects should include a detailed maintenance plan that accounts for irrigation frequency, winterizing and calibrating irrigation systems, fertilization, dealing with insect and weed infestations, and hiring a plant manager to monitor and replace plants every spring, dead head flowers and remove thatch (Griswold, 2010). Although it may be necessary to achieve certain aesthetic or production goals, intensive maintenance undermines the goals of green roofs being low-input, self-sustaining systems. Furthermore, the high costs associated with high-profile green roofs are simply not feasible for the vast majority of property owners and would be a major barrier to large scale implementation (Thompson, 2009). Even for well-funded, high-profile green roofs, intensive maintenance regimes that are prescribed may be impractical. Dvorak and Carrol (2008a) learned that unforeseen conditions proved the regimented, geometric design of the prairie-based green roof on the Chicago City Hall unsustainable. The authors point out that the green roof was initially maintained like a traditional garden, but like all ecosystems it changed over time. The original maintenance plan was slow to be implemented and unforeseen weather extremes combined with irrigation shortcomings resulted in some plants dying off while other, more appropriately adapted plants spread from their original locations and began to dominate. After two years of establishment, the roof was deemed to be disorderly and in need of taming (Dvorak and Carrol, 2008a). Site manager Kevin Carrol determined the conventional site maintenance that establishes a fixed design where plants are kept in predetermined locations to be unsustainable (Dvorak and Carrol, 2008a; Dvorak and Carrol, 2008b). They opted instead for a site stewardship approach which involves a process of reading natural systems to understand and anticipate changes over time (Dvorak and Carroll, 2008a) and is more responsive to the microhabitats of the garden and allowing plants to migrate to new locations (Dvorak, 2008b). The green roof now looks more like a rolling prairie, with seasonal progressions of blooms instead of organized drifts of plants (Dvorak and Carrol, 2008a) and proves to be very adaptable to goals of identification and maintenance of successful native and non-native species, experimentation with new species, reduction and elimination of irrigation in extensive zones and exploring best management techniques for non-herbicidal plant removal (Dvorak and Carrol, 2008b).

Beyond subjective issues of aesthetics, two potential maintenance concerns with biodynamic green roofs became clear from this study. One issue is thatch. This could present problems by effectively mulching out desired species and decreasing biodiversity, though it could be beneficial in terms of moisture retention and nutrient cycling. Designers should keep in mind that thatch could be problematic by presenting a fire hazard. However, if site-specific conditions allow, low-intensity fire could be a useful management tool to decrease thatch and fertilize soil media. Another potential maintenance concern observed in this study is the colonization of green roofs by trees and other woody vegetation. Obviously this would be problematic if the roots of such vegetation cause structural damage to the roof. Although not investigated in this study, it seems unlikely that woody vegetation would attempt to extend their roots below the drainage layer because water is not present there. One elm tree seedling established itself on the green roofs used in this experiment. The tree survived periods of drought by repeatedly shedding its leaves and seemed to be taking on a diminutive, bonsai-like form. It is possible that it could continue to survive there but never outgrow its location.

5.6.3 Native vs. Exotic Plants on Green Roofs

Even though natives were planted on the green roofs used in this study, many exotic plant species that were well adapted to green roof conditions colonized them. All of these species are widely naturalized in the surrounding areas. Given the likelihood of colonization by such species, reasons for disallowing specific species from green roofs need to be understood before drastic attempts are made to remove them. An exotic species that acts as an agent for decreasing biodiversity would be a good example; however, this could also apply to an overly aggressive native species. Furthermore, if biodiversity is a goal then a mixture of exotic and native plants can produce extremely diverse communities as exemplified by some sensitively maintained vegetable gardens (Owen, 1991). On the other hand, there is the possibility that green roofs could serve as a vector for extremely environmentally detrimental species such as cogon grass (*Imperata cylindrica*). It seems unlikely that some exotic species such as dandelions and chickweeds can be kept from colonizing green roofs without an unrealistic level of maintenance.

A commonly stated reason for using locally native plants on green roofs is to enhance sense of place, but the term "native" is rarely defined and there is an ongoing debate about what level of nativeness is appropriate (Butler et al., 2012). At one extreme, arbitrary political boundaries are often used to define the nativeness of a species. At the other extreme, only a specific plant ecotype indigenous to a specific site may be considered truly native. How then does one apply a definition of native to a man-made environment such as a green roof? In most cases, the habitat qualities of a green roof are so different than the habitat it is replacing at ground level that it would be impractical or impossible to recreate or transplant that landscape onto a roof. Therefore a logical approach is to look to local habitats that share analogous qualities with green roofs for appropriate native plant selections. Even then, the green roof habitat will be fundamentally different and so while the resulting plant community may be very similar to its analog, it may be practically impossible to exactly replicate an actual prairie or other ecosystem on a green roof. Brenneisen (2006) stated that near-natural habitats can be established on roofs, but one could argue that a novel ecosystem flourishing on a man-made surface is just as natural as any other ecosystem. Dvorak and Volder (2013) successfully established a beautiful mixture of native and exotic succulents (and one native grass) on unirrigated green roofs in south-central Texas. This plant community may be appropriate for south-central Texas in terms of sense of place, but to what degree it is appropriate for other climates is debatable. If these plants are well-adapted to urban green roof habitats that are analogous to their native habitats in the wild, then perhaps they are naturally appropriate for a novel urban ecosystem. Care should certainly be taken, though, not to introduce exotic species that could spread from green roofs. It should be noted that there are many species of succulents native to humid subtropical climates including species of the *Opuntia, Sedum* and *Manfreda*, genera which were used in Dvorak and Volder's experiment.

5.7 Limitations

As is the case with any ecological experiment, there were many unaccounted-for variables (microclimate, elevation and proximity to other ecosystems) that could potentially affect the outcomes. Therefore it is likely that this experiment would produce different results if repeated in another location. The intent of this study is to add resolution to the currently limited knowledge of the potential for growing non-succulent, native herbaceous plants on unirrigated green roofs in a humid subtropical climate. In order to identify useful information revealed by this study, it is first necessary to understand some of its major limitations. The foremost limitation of this study is that it only investigates the first year of plant establishment. Because of this limitation, this study only provides a glimpse of the trajectory of a nascent plant community. Many of the species that were present on the platforms are early successional species and it is probable that over time they would be supplanted, at least in part, by later successional species. There is also no way to know to what degree any of the successful species perpetuated themselves on the platforms without consecutive years of study. It could be that many of the colonizing species, whether present in the seed bank(s) or introduced through normal dissemination, germinated and lived for a short time but did not really establish themselves. This raises questions as to how informative the results from the measurements of biodiversity are. Certainly they would provide a clearer picture if conducted over the course of several growing seasons. Furthermore, as mentioned in Chapter II, some researchers have suggested that the diversity and cover provided by established plants may make the platforms more habitable for a broader spectrum of species (Sutton et al., 2012). This might mean that some of the unsuccessful and marginally successful species planted on the roof could exhibit more success with a greater establishment of cover.

There are also limitations of the research design. First, the selection of species planted on the platforms as plugs and seed was limited by funding and commercial availability. There were some species that were initially targeted due to drought hardiness but were unavailable for use unless collected locally by hand in the field. This is likely to be a challenge to designers wishing to use certain natives for similar projects. Also, the species that were used were not from local genetic stock. The species planted as plugs were purchased from Oxford, Pennsylvania and may have been part of that local gene pool and perhaps therefore less adapted for heat tolerance. The species planted as seed were purchased from Junction, Texas and may actually be more adapted for heat and drought tolerance but perhaps less tolerant of humidity.

Secondly, the experiment used 6 inches (15.24 cm) of growing media which is the upper limit of what is normally considered an extensive green roof (Getter and Rowe, 2006). An initial assumption was that there would be very few species that could survive on unirrigated extensive green roofs and so by using the deeper media there would be a greater chance of identifying species with potential for green roof use.

Many potentially confounding variables arose from the introduction of additive seed banks. It was at first assumed that the commercial growing media would have been sterile in terms of seeds. There are strong indications that this was not the case and this means that the control media contained one seed bank and the treatment media contained that seed bank plus another from the native soil. More information could have been gleaned from an additional treatment of pure commercial media that had been sterilized. In this case, it would have been possible to discern what species were colonizing the platforms from the immediate surroundings. Furthermore, there is no information definitively indicating that successful species unique to the treatment platforms would experience the same success in the absence of the native soil.

The location of the experimental platforms undoubtedly also affected the outcomes. In a broad context, it is important to keep in mind that this experiment was performed in East Central Mississippi. Although the results could be used as a starting point for similar research in other climates, they may only be directly applicable to a humid subtropical climate. Site specific context also likely affected outcomes. The platforms were constructed approximately 6 ft. (1.83 m) from the ground. Even atop a single story building, this height would be greater and therefore the means by which extraneous colonizing species found their way to the growing media would be affected differently. The fact that the experiment was conducted in a pastoral setting exposes the platforms to a different suite of species than would be present in an urban setting. They would also be exposed to different species of birds which may act as vectors for introducing certain seeds. Although no measurements of ambient temperatures were taken for this experiment, it is likely that they would have been higher in an urban setting where the growing media would be exposed to reflected heat or heat released by the thermal mass of buildings at night. It is also worth noting that before the plug species were planted on the platforms, they spent time sitting on a vegetated ground surface immediately adjacent to where mowing occurred. This almost certainly resulted in the introduction of some species.

Finally, it is worth mentioning that the species planted as plugs could have been planted later in the year when there was less chance of prolonged hot and dry weather. Even though the potted plug species were given water with decreasing frequency over several months prior to planting, many seemed to react to cessation of all irrigation by going into early winter dormancy. This could have affected their survival, health or cover for the following spring.

5.8 Conclusions

This study concludes that irrigation is not necessary to support non-succulent herbaceous plants on green roofs as long as appropriate selections are made and there appear to be many native species adapted to growing on unirrigated green roof conditions. Although these results were derived in the humid subtropical climate of East Central Mississippi, this finding will likely hold true for many other regions and climates as well. Local habitats that share analogous conditions of shallow, nutrient-poor soils and frequent heat and drought stress should be used as a reference for designing green roof plantings.

It was found that adding native soil to the platforms did not significantly affect the establishment or survival of the planted species nor did it significantly affect species richness, evenness or cover. While the addition of the seed bank from the native soil may have been helpful in identifying potential species for green roof applications, this practice will likely introduce unwanted exotic and/or invasive species.

This experiment demonstrated that seeding and planting plugs can be an effective means of plant establishment on unirrigated green roofs but care should be taken to plant plugs after the threat of prolonged heat and drought has passed. It is also clear that without frequent weeding, spontaneously colonizing species will likely be present on most green roofs. Because of this, unirrigated green roofs can be novel, diverse and self-perpetuating ecosystems. As was the case with this experiment, colonizing species have the potential to contribute a majority of cover on a green roof if niches are available. In a humid subtropical climate, green roofs installed without plant materials can achieve near 100% cover by the end of the first year of establishment. However, it may take many years for plant communities to stabilize. Most of the early colonizing species will be ruderal in nature and therefore may not be acceptable depending on design goals. Unirrigated green roofs will be temporally dynamic, expressing different suites of plants each season and oscillating between verdant and arid periods.

5.8.1 Specific Design Recommendations

Based solely on the results of this experiment, a prescription for planting design for unirrigated green roofs in a humid subtropical climate can be suggested. Given that grasses were significantly more hardy than forbs (sideoats grama was clearly the most reliable species) and that the provision of cover can facilitate the establishment of other plants (Sutton, 2012), planting a contiguous grid of sideoats grama plugs and overseeding with species from tables 4.1, 5.1, 5.2 and 5.3 should produce a reliable meadow effect. The grid of sideoats grama can provide a skeleton that maximizes order and unity during periods of visual decline and create a reliable source of cover, allowing other species to establish themselves in the spaces between clumps. Little bluestem (*Schizachyrium scopirum*) and purple love grass (*Eragrostis spectabilis*) were also successful and could be incorporated into the grid to create visual elements such as drifts and other patterns. However, these two species were established from seed and their rate of survival when planted as plugs is unknown from this experiment. Regardless, the inevitable plugs of grass that die could be replaced or the vacant niche could be allowed to fill in which would add a dimension of unexpectedness to the design.

The plugged species used in this experiment were planted in a grid, 12 inches (30.5 cm) on center. Using a smaller grid distance will increase order and thus temper the chaotic appearance of early successional species by decreasing the available niches for these plants to flourish unchecked. Varying the grid distance within a design could also be used to create visual elements. Spontaneous colonization of unplanted soil media should be expected. Areas where plants are not desired could be covered with pavers or flat stones.

5.8.2 Future Research

This experiment can be used as a template for future research aimed at discovering plant species that are adapted to survive on green roofs without irrigation. For humid subtropical climates, the list of successful species presented in this paper should be expounded upon and examined at shallower soil media depths. Local habitats that share analogous qualities with green roofs should be examined for candidate species.

This study also presents a method to quantify the biodiversity of plant communities on green roofs and other man-made habitats. The biodiversity of existing green roof plant communities that have been allowed to stabilize over time should be measured. These measurements can be used to rank green roofs against other habitats in terms of biodiversity in order to more clearly understand their potential contribution to sustainable development. With regard to policy, biodiversity performance standards could be used to incentivize green roof installation.

Combining native soil with green roof media in the hopes that microbial inoculation will aid in plant establishment appeared from this study to be less fruitful considering there are many plants well enough adapted to surviving on green roofs. Soil seed bank sampling can be a useful research tool to identify plants for green roof use but ruderal species are more likely to benefit from this. Obviously care should be taken not to disturb sensitive habitats. The soil samples used in this experiment were taken from disturbed sites which resulted in the establishment of many exotic species. This practice also carries with it a risk of introducing a truly noxious species and for this reason is not recommended for actual installations. However, inoculation with native soil probably has the potential to introduce beneficial species that would not otherwise find their way onto green roofs including plants, but also microbes, fungi, cryptogams and animals.

Lastly, it was observed in this experiment that even though yellow coneflower (*Ratibida pinnata*) experienced high mortality rates, the few individuals that survived, exhibited excellent health throughout the experiment. This may indicate a degree of variability within the taxa that would allow for developing more drought resistant varieties or cultivars specifically for green roof use. Desirable species that are only marginally successful on green roofs could be investigated in this capacity.

CHAPTER VI

THE PRAIRIE GREEN ROOF AS A MODEL

The results of this study disprove what seems to be a prevailing perception among green roof researchers and professionals; that non-succulent native herbaceous plants cannot survive on green roofs without irrigation and other regular human interventions. This experiment demonstrates that there is a broad range of herbaceous plants that are adapted not only to survive, but thrive under unirrigated green roof conditions, making the prairie green roof model viable and likely the most practical option, at least for humid sub-tropical climates. Monterusso, Rowe and Rugh's influential 2005 study seems to have influenced many researchers to prematurely conclude that this is not possible. However, Monterusso, Rowe and Rugh used a relatively narrow selection of prairie natives for their experiment, mostly limited to species that are commonly available in the plant trade. Their study probably would have yielded different results if more specialized species, specifically those indigenous to truly xeric habitats such as glades, barrens or outcrops, had been targeted for study. Species from these habitats are pre-adapted to dry, nutrient poor conditions. They also exclusively planted plugs which were likely grown in heavily irrigated nursery conditions, thus making a transition to drought conditions more difficult. Monterusso, Rowe and Rugh attempted to wean the plants from irrigation through a prolonged establishment period but still experienced the majority of their mortalities after the termination of irrigation. This experiment demonstrated that better results can be gained from sowing seeds directly onto the soil media. This practice introduces a much larger sample of genetic combinations and increases the probability of having individuals of a certain species find success.

Experiments and case studies regarding prairie green roofs represented in the literature almost ubiquitously use an anthropocentric paradigm of regular irrigation, fertilization and opposing natural succession through disallowance of unintended species. Green roofs become less sustainable (in terms of input/benefit ratio) and less accessible to potential implementers (in terms of cost) as required maintenance is intensified. More importantly though, these practices all work to negate many of the benefits that green roofs can offer.

Water holding capacity, the primary intended function of green roofs, is decreased when irrigated. Use of potable water for irrigation is unsustainable and defeats goals of water conservation. Furthermore, regular irrigation favors plants that are poorly adapted to living on green roofs, thus reinforcing the need for irrigation, but it also increases the likelihood of domination by one or a few species, thus limiting biodiversity. Fertilization of green roofs not only needlessly increases cost, but also decreases the quality of effluent from the roof and is counterproductive in helping plants adapt to green roof conditions. Natural nutrient cycling should be a design goal for green roofs that are expected to be low-input, self-sustaining systems.

The potential for green roofs to act as reservoirs of biodiversity is possibly the greatest benefit that roof greening offers. This cannot be achieved if a narrow plant list is too strictly adhered. Changes in plant community composition through natural selection is going to occur by default in the absence of human intervention and for roofs to attain high levels of species richness, like the Moos water filtration plant (Landolt, 2001), these changes must be allowed to occur. Succession will include an accumulation of unexpected species and shifts in abundance of planned species. As plant matter decomposes and the green roof soil media accumulates a broader range of microbes, fungi and other biota, soil formation occurs. As soil formation occurs, the green roof is more habitable to a broader range of organisms including more specialist and rare species. These changes result in the more ruderal early successional species becoming less frequent. As evidenced by the Chicago City Hall prairie green roof (Dvorak and Carrol, 2008a), an impractical level of maintenance is required to keep a grassland plant community in stasis. It is also unrealistic for designers to attempt to transpose expectations of conventional, ground-level landscape beauty onto a medium that more closely resembles a desert in terms of hydrologic function and aesthetic range. Instead, this is a challenge that must be met by designing with an ecological approach. To design ecologically is to understand that natural systems are fluid and change over time. There is a near infinite amount of variables affecting such systems that designers cannot possibly foresee and therefore our interventions only serve to guide or retard processes that are always occurring and mostly invisible to our senses.

In order for green roofs to provide the ecosystem services of significant stormwater management and heat-island effect amelioration, vast areas of urban rooftops must be greened. At this scale, it is also possible to create large enough expanses of high-quality grassland habitat as to have a significant ecological impact. This is all the more important as the remaining natural patches of this habitat become increasingly rare due to agricultural expansion and urban sprawl. Vast areas of billowing meadows adorning urban rooftops would not only be beautiful to behold, but provide refuge for innumerable species that include some that are very sensitive to disturbance. Because green roofs provide a harsher growing environment in flux between greater environmental extremes, large scale green roof habitat could facilitate expedited adaptation to global climate change through natural selection.

As explained in chapter 2.2.2 and 5.6.1, watershed-scale roof greening efforts should be focused on the expansive flat roofs of commercial and industrial areas for maximum effectiveness. Most of these roofs are already built and would have to be retrofit. As J. William Thompson (2009) points out, green roofs as high-cost, boutique design elements are simply not a viable model for most roof greening projects at this scale. Design professionals must be prepared to offer a cheap, effective and self-sustaining green roof option. This study demonstrates that biodynamic prairie roofs can fulfill this role. The following is a list of unirrigated prairie green roof installation and maintenance strategies ranging from least to most input required. See Chapter 5.8.2 for specific design recommendations.

Option 1: Install a layer of soil media only and allow it to be spontaneously colonized by herbaceous plants. As suggested by the results of this experiment, green roofs installed without any plant materials may possibly achieve near 100% cover by the end of the first growing season. Always make design and maintenance provisions for HVAC infrastructure and other places where vegetation is not desired. The roof should be inspected once per year and any colonizing woody vegetation removed.

Option 2: Install a layer of soil media along with a seed mixture of appropriate prairie species. See tables 4.1(p.g. 47), 5.1(p.g. 70), 5.2(p.g. 71), and 5.3(p.g. 73) for a list of recommended species to try for a humid subtropical climate. For other climates, nearby xeric habitats should be examined for possible candidates.

Option 3: Install a layer of soil media, prairie seed mixture and grass plugs. To avoid irrigation, grass plugs should be already established in green roof soil media and planted when dormant. Otherwise some amount of irrigation must be provided between the time of planting and their first dormancy. An initial watering is still advised immediately after planting. See chapter 5.8.2 for more information on using grass plugs.

Option 4: For highly stylized green roof installations, spontaneous colonization, prairie seed mixtures and grass plugs can all be used to create planting compositions. Making use of varied soil media depths, paths, cover objects and sculptural elements can create legible patterns. See chapter 5.6.1 for further information on aesthetics.

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