A Study in the Comparative Viability of Green Roofs Constructed Using Native Accent Species Relative to Green Roofs Using Sedum Accent Species: A First Step Toward the Potential Development of Green Roofs as a Tool for Creating a Transition Zone Between Native and Urban Ecosystems

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

Modern urban development over the last 200 years has led to the construction of cities which create severe issues for native ecology, human health, and the safety of both people and property. Among these issues are the urban heat island effect, flash flooding due to paving of stormwater runoff paths, and urban ecosystems acting as disruptions to surrounding native ecosystems. Green roofs help mitigate flash flooding and the urban heat island effect, and they have the potential to be developed into a tool for integrating urban and native ecology. Because green roofs can have a lifespan of thirty years or more and due the high initial set up cost of a full green roof, this project functions as a proof of concept for a long-term study that might advance the development of such a tool, while minimizing the loss if one or more of the green roof designs is completely non-viable. There were also plans to assess the impact the accent communities had on ecosystem services, however disruption due to Covid-19 restrictions made this impossible. This project found that green roofs planted with a community of experimental native accent species established themselves successfully to the same degree as those planted either with proven native accent species or sedum accent species, and the accent community present had no bearing on the establishment of the sedum groundcover communities, with neither Blocks B or C showing statistically significant change in vegetation coverage year over year that could be attributed to either irrigation treatment or accent community. Furthermore, after one year, neither irrigation treatment nor accent community had a significant impact on the degree of vegetation cover. Based on these results, the green roof designs used in this project are viable enough at least for use in a long-term study to test the ability of the design to actually facilitate the integration of urban and native ecosystems without the risk of early green roof failure.

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Table of Contents

List of Figures

List of Tables

1. Synthesis of Literature on How Greenroofs can Contribute to Positive Environmental Change

Modern development of cities and suburban areas disrupts the ecology of native ecosystems and the broader function of environmental systems, cuts residents off from green space, and exacerbates anthropogenic environmental problems in ways that are detrimental to human comfort, harmful to human health and safety, and wholly unsustainable. Two major examples of these effects are the urban heat island effect and flash flooding due to stormwater runoff. One tool that has shown potential to help mitigate the urban heat island effect and reduce stormwater runoff is modern green roof technology. Green roof technology is an alternative to traditional roofing that replaces the outer layer of a conventional roof system with a vegetation bed.

Modern urban design creates cities that disrupt the continuity of native ecosystems. In areas that were once prairies, coastlines, or grassland, modern inner-city areas stand as artificial mountains, while suburbs act as new rocky deserts, and both are incompatible with the native ecosystems they replace. These words are more than flowery language. Michael Johnson (2014) identifies several environmental impacts of urban sprawl, including loss of environmentally fragile lands, reduced regional open space, greater air pollution, higher energy consumption, increased runoff from stormwater, and increased risk of pluvial flooding, or flooding caused by rainfall (2001). The issue of stormwater building up in paved areas of cities and the need to move that runoff elsewhere has been well understood since the construction of humanity’s first major cities, as evidenced by the sewer drainage systems found in every developed city (Sedlak, 2014). A perfect example of this phenomenon is the case study of the Speedvale Experimental Basin near Guelph, Ontario, which was developed into an urban watershed between 1975 and 1982 (Cook and Dickinson, 1986). The study found that though there was no statistically significant change in mean annual rainfall, there was a statistically significant increase in mean annual runoff discharge, which can be interpreted as an increased frequency of pluvial flood events. More importantly, the mean annual runoff coefficient, or the ratio of annual runoff volume relative to annual rainfall, increased by a factor of 1.5, but the mean peak flood volume increased by a factor of 2.9, indicating that flood intensity is more sensitive to urbanization than total runoff. To summarize, the study found that urbanization and increase in paved surface area not only increases the frequency of pluvial flood events by 50%, but it also produces a near three-fold increase in the severity and potential damage of pluvial flood events. Beyond monetary damage from floods, increased flooding in urbanized areas is made worse by combined sewage overflow systems (CSOs). In a combined sewage overflow system, sewage and wastewater flow into the same main sewer system as the storm drains. As a result, when storm systems get overwhelmed during major flood events and the overflow systems release, untreated sanitary sewage is released into the environment. Untreated sanitary sewage is the outflow from plumbing systems being transported to water treatment facilities. Between 2013 and 2019, nearly 1.04 billion m^3 of sewage and wastewater effluent was reported as being released by CSOs in Canada, including over 122 million m^3 reportedly discharged in Ontario municipalities alone according to the Effluent Regulatory Reporting Information System. As Sedlak (2014) points out, simply replacing CSOs with separate sewage and stormwater systems is a monumental and prohibitively expensive task. The process would also be incredibly time consuming. The city of Cincinnati spent $1.01 billion US between 2009 and 2021 to complete the first of two phases in an effort to remediate its CSO problem (Metropolitan Sewer District of Greater Cincinnati, 2010; 2023).

A second major negative impact of modern urban design is the urban heat island (UHI) effect. First documented in 1833, the urban heat island effect is a phenomenon wherein the urban interior of a city is warmer than the surrounding areas (Oke, 1982). This UHI effect raises air temperatures by 5˚-15˚C (Mohajerani et al, 2017). According to Mohajerani et al, the primary causes of this increase relate to the effect loss of vegetation, prevalence of low albedo surfaces, and anthropogenic heat sources have on total heat storage. Vegetation holds water in the soil while also releasing water through evapotranspiration. The water in the soil and vegetation acts as a heat sink, increasing the amount of energy required to raise temperatures with a minimal effect on volume, while the evapotranspiration of vegetation decreases total stored heat. Albedo is a measure of the reflectivity of a surface, meaning that low albedo surfaces reflect less sunlight than high albedo surfaces. As a result, low albedo surfaces absorb more of the energy from solar radiation. The replacement of vegetation and soil with low albedo surfaces like asphalt and conventional roofing materials therefore removes a major energy sink along with a mechanism for heat loss and replaces it with material that easily absorbs energy from the sun and rereleases it as heat. This coupled with anthropogenic heat sources like vehicles, industry, and human habitation creates what amounts to a bubble of built-up heat enveloping urban areas. While the UHI effect can be observed most of the time, it is most obvious during times when that heat can be released (Tam et al, 2015). It keeps urban areas warmer during the Winter and, during the Summer, prevents urban areas from being able to cool down at night.

Increase in temperatures both on average and during Winters and nights creates a number of issues for human health. A number of studies found a link between the urban heat island effect and mortality during excessive heat events. A study conducted in Wuhan City used UTCI, or Universal Thermal Climate Index, as a metric for heat stress (Dong et al, 2020). UTCI is a calculation for effective temperature for heat exchange between the human body and the environment. The study found that the average UTCI in the main urban area was 35.05˚C, 0.69˚C higher than the average UTCI found outside the urban area. A study of the August 2003 heatwave in Paris found that nighttime temperatures between urban and suburban areas varied significantly and were significantly correlated with deaths of individuals aged 65 and older between the regions, while daytime temperatures between urban and suburban areas did not vary significantly (Laaidi et al, 2012). A review published in the Canadian Medical Association Journal lists age as the primary factor in mortality due to heat stress, but also lists cardiovascular diseases, respiratory diseases, and type I and type II diabetes as conditions that can increase an individual's chances of death induced by heat stress (Kenny et al, 2010). Additionally, a study by Obradovich et al (2017) determined a link between higher nighttime temperatures and a loss of sleep, which carries its own host of detrimental health effects.

In the face of more frequent and extreme heat waves, such as the 2021 heatwaves that hit Portland and Vancouver, urban areas must be redesigned to mitigate the urban heat island effect. The Vancouver heat wave killed at least 585 people across British Columbia and was described by news sources as the deadliest weather event in Canadian history (Schmunk, 2021). As detailed above, loss of vegetation and loss of the heat sink provided by water trapped in soil is one of the major contributors to the urban heat island effect. Coincidentally, loss of soil bed and vegetation water storage capacity is one of the major contributors to flash flooding. Both issues are also impacted by the prevalence of paved surfaces; the low albedo of paved surfaces increase heat absorption and their impermeable nature drives stormwater toward flood zones. Therefore, one solution to help mitigate both issues is the replacement of paved surfaces with urban green space, areas of soil and vegetation. The problem with this solution is the issue of location for these green spaces. Paved roads remain integral to the logistics of the modern world, and tearing down half of the buildings in urban areas to replace them with empty plots is simply not viable. The question becomes how to have green space and buildings in the same place, and the answer is the modern green roof.

A multitude of studies have proven that green roofs can be used to help mitigate the UHI effect. One such study used modeling of the January 2009 heatwave in Melbourne, Australia, to assess the UHI impact (Imran et al, 2018). By comparing traditional and green roof surface temperatures to rural areas around Melbourne, the study determined that the temperature increase due to UHI was decreased by 1˚C at 30% green roof cover and by 3.8˚C at 90% green roof cover. Furthermore, green roofs also decreased UTCI by 1.5˚C and ground level and 5.7˚C at roof level. Equally well established is the capacity of green roof systems to aid in mitigation of stormwater runoff. There is general consensus that green roofs installed over a larger area allow for soil capture of stormwater to delay peak flood times and decrease peak flood volume. Several studies, including Zhang et al. (2015), have studied the impact green roof systems have on pollutant loads in green roof runoff. The green roof observed in the Zhang study showed an annual total rainfall retention of over 750mm and an annual average retention rate of 68%. The study determined that substrate depth, vegetation cover, roof age, and slope all impact stormwater retention, and by comparing their work to a Connecticut study from 2011, a UK study from 2009, and a North Carolina study from 2008 noticed that greater soil depth increases stormwater retention (2015).

Capacity for stormwater retention and UHI mitigation are two examples of the functional green roof benefits referred to as ecosystem services. Another prominent ecosystem service is reduction in building energy consumption. In the Toronto region, air cooling was found to scale with surface cooling, and air cooling efficiency of green roofs was greater in green roofs populated by plants in the genus *Sedum* than in those populated with meadow plants (MacIvor et al 2016). A study in Hong Kong found that older buildings with outdated insulation in particular experienced a decrease in heat transfer (Jim, C. Y., 2014). Susca (2019) conducted a literature review found that the insulation benefit varied across warmer climates, but there was a lack of study on colder climates. Susca (2019) did determine that in a LEED certified building in Toronto, the building insulation was robust enough that the presence of a green roof had no impact. Two different models in a probabilistic study found that on a 2000 m^2 roof, the green roof lowered energy costs, but one model calculated the savings at $710 per year, while the other calculated the savings at $1670 (Clark et al, 2008). The reduced energy usage combined with the improved insulation capacity also contributes to a decrease in anthropogenic heat. Because anthropogenic heat is a contributing factor in the UHI effect, green roofs constructed across a large area have the potential to produce a positive feedback loop that amplifies the capacity for UHI effect mitigation.

UHI effect mitigation, urban energy conservation, and stormwater retention are the most concrete and well documented of the ecosystem services green roofs provide, and can be viewed as the tangible, measurable, intrinsic ecosystem services, for which a dollar valuation of the benefit can be calculated, and from which a direct positive impact on human health and safety can be observed. There are, however, a host of other ecosystem services green roofs have the potential to provide. These additional ecosystem services can be contextualized for the purposes of this thesis as either environmental benefits or human quality-of-life benefits. These benefits are not intrinsic impacts of green roofs, and each benefit's magnitude is dependent on whether or not providing that impact was taken into consideration and pursued as a benefit in a given green roof's design. The environmental benefits are the element of green roofs that the core theoretical framework of this project centers on, but this thesis will take the time here to briefly touch on human quality-of-life benefits, simply to identify them and explain from a purely philosophical standpoint why they are important to understand but not relevant to the work. A green roof's human quality of life benefits are its aesthetic value, the value of the green space as a physical amenity, and the value of the positive impact green space has on mental health. All of these benefits are highly subjective, and the values are difficult to quantify, particularly the mental health impact due to the fact that mental health is a catch all concept that includes mental illness and psychological distress as well as other factors. As previously stated, the quality-of-life benefits must be intentionally accounted for in the design of the green roof for them to be provided. This can result in a situation where quality-of-life benefits compete with environmental benefits for priority, resulting in a green roof that instead of focusing on one set of benefits or the other simply settles for being ineffectual at providing either set of benefits. The quality-of-life benefits are still valuable and important, however, because firstly, increasing human quality-of-life is an inherently good thing, and secondly, the quality-of-life benefits can be a driving force for green roof adoption on a personal level. In fact, what we consider the quality-of-life benefits of green roofs were probably a major factor in the original development of modern green roofs.

As stated before, what this thesis will refer to as environmental benefits are the core of the theoretical framework. These benefits are measurable impacts of green roofs on the biological ecosystem, water quality, and air quality that can be taken into consideration and prioritized for during the green roof design phase. Airborne particulate pollution capture, carbon sequestration, and the provision of habitat for arthropod communities are all examples of environmental benefits. Research has shown, however, that the current design framework for green roofs is insufficient for use as a tool for aiding in restoration and reintroduction of historically native species to urban ecosystems (MacIvor et al 2011).

An ongoing study by Stephen Murphy (my advisor, pers. comm.) is indicating that smaller tallgrass prairie restoration sites in cities do not perform as well as larger sites outside urban areas, and has identified lack of internal area and/or external interconnectedness necessary to maintain the required pollinator communities. This is not to say that the urban restorations are without merit as other studies have indicated that they can be successful compared to degraded sites (Stoner and Joern 2004; Leston and Koper 2016) – it simply means that success is a relative term that requires careful definition and managed expectations.

Since 2009, major cities like Toronto, Ontario, New York City, Portland, Oregon, and Gatineau, Quebec have implemented bylaws requiring green roofs. An increased demand in cities for urban green space in the form of public parks and green roofs, when coupled with a rising interest in replacing traditional cultivated lawns with native species, presents a unique opportunity to address this lack of external interconnectedness (e.g. Smith and Fellowes 2014; Zheng and He 2021). Picture the mass transit system in southern Ontario. The Via Rail and GO Bus systems connect major stations in the major cities and city clusters. These cities and metropolitan areas in turn have their own mass transit systems with major stations, smaller unofficial stations where multiple bus, subway, and light rail lines intersect, and bus stops along the individual lines. The purpose of this set-up is, obviously, to connect neighborhoods to their closest city centers, then connect those city centers together into city-wide systems, and finally connect those cities into a unified region. Compare larger restoration sites to the major metropolitan areas, and the smaller restoration sites to the population centers within those metropolitan areas. These sites are too far apart for their pollinator communities to effectively interact. The strategic placement of urban green space like city parks and green roofs seeded with plants favorable to these pollinator communities could theoretically create a network unifying these sites into a single interconnected pollinator and plant community.

Designing this system would be a major undertaking, and implementation would require coordination between city planners, the horticultural industry, and private businesses and citizens, but the only major technological barrier is the development of a green roof system capable of serving as a functional component of such a system without the need for cost prohibitive maintenance. Such a green roof would need to be able to support plant species native to the chosen ecosystem type, or at least serve as a functional habitat for the generalist pollinators that specialist plants native to the selected ecosystem rely on. It would need to be able to, at minimum, retain its plant community and at best develop its own viable seed bank. Because green roofs are expected to last as long as 30-50 years, a full study on the long-term viability of such a green roof system would need to span a minimum of 5-10 years, assess viability and efficacy at multiple building heights, measure the impact of various soil blends, and assess the effect soil blends and plant communities have on established ecosystem services and stormwater runoff quality. Likely this would mean finding multiple partners already planning to construct green roofs who are willing to host these experimental designs. The US EPA reports that extensive green roofs cost an average of around 108 USD per square meter just to install, and that number jumps to around 324 USD per square meter for intensive green roofs. As this figure doesn't account for the increased cost of sourcing native plants or the cost of any maintenance, any partners would need assurances that the green roofs wouldn't immediately die and need replacing the next year. The intention of this project, therefore, is to function as a proof of concept in preparation for such a long-term study. The next section of this chapter will focus on green roof design considerations, discuss how the proof of concept will be set up relative to a hypothetical design for the later study, and identify what questions will be addressed by the proof of concept in preparation for the later study. So what is a modern green roof?

Sod roofs have been used by Germanic people for centuries, even in the failed Viking settlements of Newfoundland and Nova Scotia, and rooftop gardens date all the way back to Mesopotamia. Records of trees being purchased for ancient rooftop gardens a present in the writings of Pliny the Elder. The modern green roof, however, originated with Reinhard Bornkamm, whose work on gravel-covered roofs laid the framework for the research that would lead to the 1969 construction of the Geno Haus, the first modern green roof. As the basic design elements of green roofs are well understood, this section is a synthesis of information from "Green Roof Plants: a Resource and Planning Guide," by Edmund and Lucie Snodgrass (2006), "The Green Roof Manual: a Professional Guide to Design, Installation, and Maintenance," by Edmund Snodgrass and Linda McIntyre (2010), and "Green Roof Construction and Maintenance," by Kelly Luckett (2009). The general construction of a green roof is similar to that of a conventional roof, with the structure and the insulation and waterproofing layers being the same in both systems (Luckett, 2009). There are a multitude of ways to set up each part of a modern green roof, but all elements should comply with the recommendations set forth by the Forschungsgesellschaft Landschaftsentwicklung Landschaftsbau, or Landscape Research, Development, and Construction Society (FLL), the German organization that has taken on the responsibility of developing detailed guidelines for planning and construction of green roofs. While the structure generally must be able to support a higher load, the primary difference is that in a green roof, the outermost layer is replaced with a containment system filled with a substrate called a growing medium, as well as a vegetation layer. The containment system includes a drainage system that is either part of the container or a separate layer under the container, and it may or may not include an irrigation system. Design and construction of these layers varies widely between green roofs based on the scale and purpose of the green roof. The FLL Guidelines for the Planning, Execution, and Upkeep of Green Roof Sites is a set of standards for green roofs published by the Landscape Research, Development, and Construction Society in Germany (Snodgrass and Snodgrass, 2006). These guidelines are considered the closest thing to an industry standard for green roof construction, and they provide a comprehensive list of parameters for all components used in the construction of green roofs. Green roofs are generally divided into two categories, extensive green and intensive. Extensive green roofs have a maximum substrate depth between 15cm and 20 cm, depending on the authority consulted (Wilkinson et al 2014). Intensive green roofs are generally defined as any green roof with a substrate depth too great to be classified as extensive, though some sources classify a green roof with a substrate depth between 15 cm and 20 cm as "semi-intensive." Because an intensive green roof with a substrate depth of 15 cm can have a load of 282 kg/m^2 at maximum water retention, and the average conventional roof has a load of 98 kg/m^2, retrofitted green roofs almost exclusively use extensive green roofs. Extensive green roofs made up more than 80% of green roofs in Germany in 2008, and though specific aggregate green roof data isn't readily available for North America, the majority of green roofs built in North America are extensive green roofs as a result of lower maintenance requirements, lower construction costs, and the practical capacity for extensive green roofs to be built as retrofit roofs (Snodgrass and McIntyre, 2010).

There are two main methods of green roof construction. The first is modular construction, and the second is build-in-place construction. Modular green roofs use trays that are made out of rigid, durable plastics. Modules usually hold 15 cm of substrate or less, with most holding no more than 12 cm. They are able to be pre-grown in a greenhouse and installed individually on a green roof. They also have the drainage system built in and act as their own root penetration barriers. With build-in-place construction, the container is built into the roof. The drainage system and root barrier are placed in the bottom of the containment area, and the growing medium is added on top of that. Modular and build-in-place construction each have their own advantages and disadvantages. Because modules can be pre-grown in an off-site greenhouse, odds of establishment success increase, and extra modules can be prepared in order to ensure that only modules that fully establish will be planted. Modules also allow for easier maintenance, as they can be moved after installation.

However, modules also impose a functional maximum depth on a green roof. This is because modules must be moved fully constructed, and increasing the depth also increases the weight of the module. This limitation restricts the kinds of plants that can be grown on the green roof. There is also the question of whether the disjointed nature of modular green roofs amplifies edge effects. With build-in-place green roofs, the depth of the substrate is only limited by the load the structure is designed to support. This allows deeper-rooted plants to be grown, making build-in-place construction preferable for intensive green roofs. The main drawback of build-in-place construction is that no maintenance can be done without directly disturbing the green roof vegetation.

Currently, green roofs are constructed using a narrow grouping of plant species. Plants used in green roofs fall in to one of two categories based on their functional role. The first category is the ground cover species. Ground cover species primarily grow low and spread out wide. Their role is to maximize two-dimensional cover and grow root networks to secure the substrate in place. The tables in Appendices A and B detail a tabulation of the plant species listed in the book “Green Roof Plants” by Snodgrass and Snodgrass (2006). Appendix A lists genera used for groundcover species and Appendix B lists genera used as accent species. Fifty of the 89 groundcover species are in the family Crassulaceae, with 37 of those 50 being in the genus *Sedum* (2006). *Sedum* is also the only genus that is planted as a vegetative mat, and *Sedum* and *Delosperma* are the only genera that can be planted as cuttings. The second group is the accent species. These species are planted for aesthetic value rather than specified functional purposes. As a result, there is a much greater variety of genera used as accents. Of the 219 species listed by “Green Roof Plants,” 127 are accent species. The genus *Sedum* also makes up 22 of the accent species. Making up 41.6% of the groundcover species and 17.3% of the accent species, the genus *Sedum* has become ubiquitous in green roof construction. This is especially true in green roof research, where nearly every study that does not specifically looking at the impact of plant structure on the project uses sedum as a default. With this information in mind, it stands to reason that a green roof design that uses native species as accents while maintaining the use of sedum as groundcover should not perform differently relative to a green roof using traditional accents, so long as the native species used are able to withstand the unique stress factors of a green roof.

Snodgrass and McIntyre (2010) identified four main methods of green roof planting: cuttings, plugs, mats, or seeds. Snodgrass and Snodgrass (2006) discourages use of seeds exclusively, however, because the time seed would take to establish leaves the green roof vulnerable to soil erosion. Vegetative mat planting is the most common method. It involves large mats of a small number of species being unrolled fully grown over the growing medium. This method is severely limited by the small variety of species available but is the fastest way to fully cover a new green roof. Vegetative cutting planting is a direct sow method similar to seed planting, but plants are able to establish faster, and are not limited by the question of seed viability. Cuttings are species limited like vegetative mats, however, because species planted as cuttings must be green roof suitable and capable of vegetative propagation. Plugs are whole juvenile plants placed into the growing medium. Plugs give the greatest variety of viable species, and they have the highest success rate. Plug planting can limit vegetation cover, however, and planting plugs closer together ceases to increase cover beyond a density of 40 plugs per square meter.

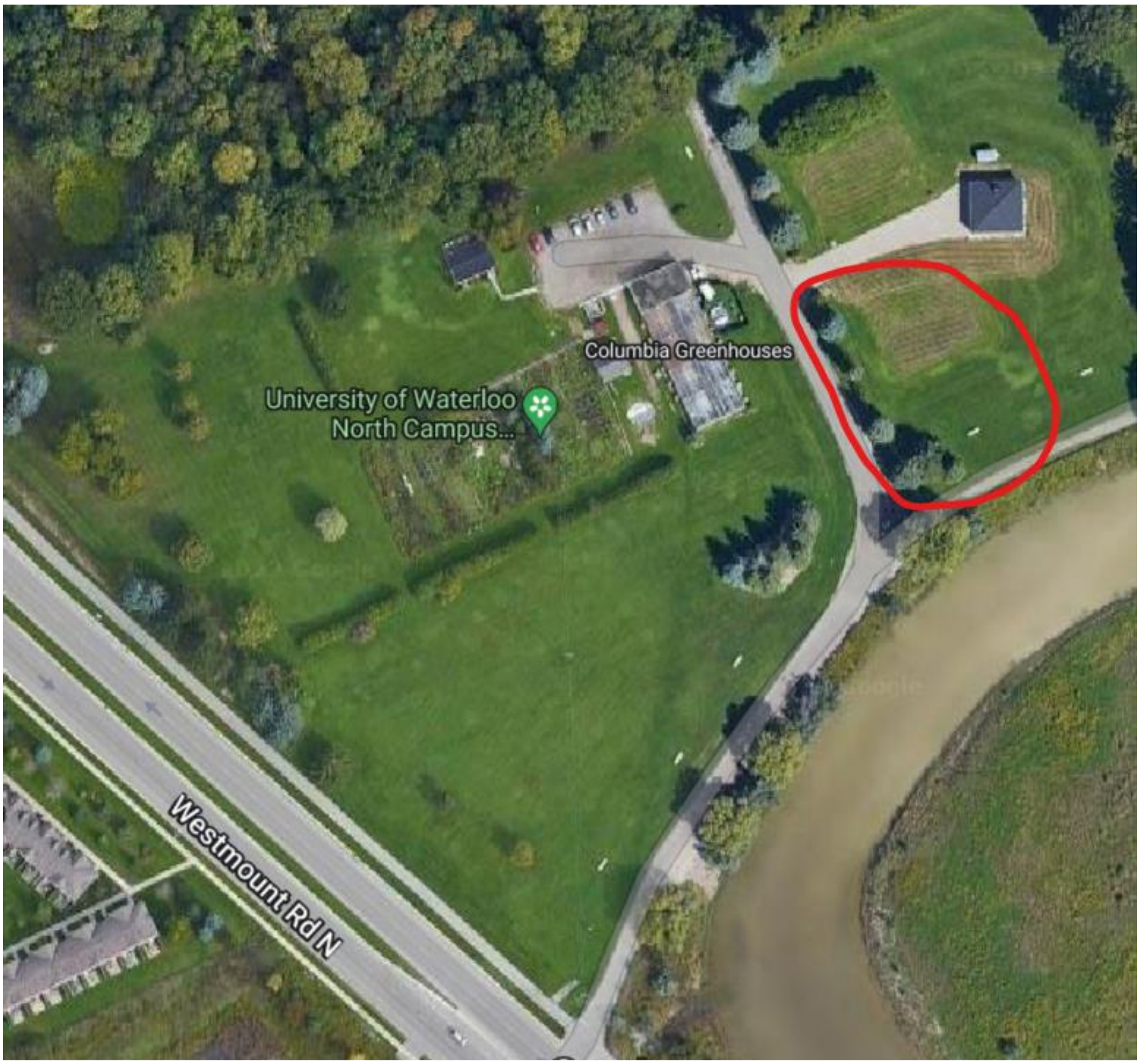
The overarching goal of this thesis was to act as a proof-of-concept study for a larger study with the intent to develop green roofs which could maintain their capacity to mitigate urban flash flooding and the urban heat island effect while also functioning as tools to aid in urban ecological restoration. This proof-of-concept study was designed around three major questions. Firstly, how effectively do green roofs with native accent species establish themselves relative to green roofs planted with exclusively sedum species? Secondly, how reliably do native accent green roofs survive the first year overwintering relative to green roofs planted with exclusively sedum species? Thirdly, how does reduced frequency of irrigation impact vegetation cover in green roofs planted with native accent species compared to green roofs planted with exclusively sedum species? Based on these three questions, three accent groups were selected. The first was an accent community made up of *Sedum likadense*, *Sedum ochroleucum*, and

Sedum telephium Emperor’s Wave, all sedum species recommended in “Green Roof Plants.” The second accent community was made up of *Allium cernuum* and *Bouteloua curtipendula*, both native species recommended in “Green Roof Plants,” sourced from St. Williams Nursery and Ecology Centre. The third accent community included *A. cernuum* and *B. curtipendula*, along with *Aquilegia canadensis*, *Penstemon digitalis*, *Penstemon hirsutus*, and *Sporobolus compositus*. These four additional species are not recommended in “Green Roof Plants,” but are native species in the same genuses as other species the book recommends, and each one has traits that theoretically make them suitable for use on green roofs. These species were also sourced from St. Williams Nursery and Ecology Centre. The plots used in this proof-of-concept study were modules with a total depth of 200 mm and a substrate depth of 150 mm which were placed at ground level for convenience and safety reasons, but in a long-term study, rooftop level sites constructed in place to limit the impact of edge effect and better mimic a natural ecosystem. While not one of the initial questions, a final goal of this project is a post-study analysis of what worked and what was learned that might impact the design of a future study.

2. Experimental Design, Analysis, and Conclusion

2.1 Site Design

The study site was set up at the Columbia Lake Ecological Reserve due to the relative isolation and the ease with which it could be accessed without a vehicle. The site appears on the campus map here under the code COG. The red circle on the map of the site below shows the location where the project was set up.



The original site construction contained three identical blocks of study modules. The diagram below depicts the layout used for a single block. As shown, each block contained six columns of five modules each, with a 50cm gap placed every two columns to allow access to all modules for analysis. The T1-T9 labels identify which of the nine treatment combinations was present in each module.

T1, T2, and T3 groups were planted with known native accent species. T4, T5, and T6 groups were planted with experimental native accent species. T7, T8, and T9 groups were planted with sedum accents. T1, T4, and T7 groups were watered every four weeks after establishment. T2, T5, and T8 groups were watered every two weeks after establishment. T3, T6, and T9 groups were watered every week after establishment.

T0 refers to squares of traditional roofing that were originally intended to be constructed and used in ecosystem service comparison studies, but these were never constructed due to limitations imposed by the Covid-19 lockdown.

T2	T5		T1	T7		T8	T5
T3	T7		T8	T5		T3	T3
T6	T9		T2	T4		T7	T2
T9	T9		T8	T1		T0	T0
T6	T6		T0	T4		T1	T4

Figure 1. Diagram of a Single Block

Module Construction

The modules were constructed in Leiyuan Greening Solution GF520 500x500x200 mm green roof trays. These trays were chosen because only two green roof tray models with the necessary depth of 150 mm were identified, and while the more expensive US-made trays would have lasted longer, the project was only intended to last a year, and doubling the cost of the trays was deemed unnecessary. Each tray was filled with 0.05 m^3 of of Earthco Soil Mixtures extensive green roof growing medium. This growing medium was selected based on cost, the fact that it was specially formulated for use on a green roof, and the fact that the provider was willing to sell it in the relatively small volume needed for this project. The modules were set up on June 27th, 2019.

All modules were planted with *Sedum spurium*, *Sedum sexangular*, and *Sedum kamtschaticum* as groundcover species, because these species are recognized as suitable for use as green roof groundcover for this region and because I have experience with these species. *Sedum album* was intended to be used as a fourth groundcover species, but the obtained specimens did not survive long enough for the green roof modules to be planted. To make up for this, each module in Block A was planted with 2 plugs of *S. Spurium*, each module in Block B was planted with 2 plugs of *S. sexangular*, and each module in Block C was planted with 2 plugs of *S. kamtschaticum*. This was not expected to affect the outcomes in any significant way, as each block was to be analyzed separately.

Each module was planted with six accent plugs based on the treatment. Sedum accent treatment modules received two plugs each of *S. likadense, S. ochroleucum,* and *S. telephium* Emperor's Wave. Established native accent treatment modules received three plugs each of *Allium cernuum* and *Bouteloua curtipendula*. Experimental native accent treatment modules receiving one plug each of *Allium cernuum* and *Bouteloua curtipendula* along with *Aquilegia canadensis*, *Penstemon digitalis*, *Penstemon hirsutus*, and *Sporobolus compositus*. Original plans called for the experimental native treatment to include *Sporobolus cryptandrus* and *Carex eburnea*, but both of these species failed in the nursery, and *P. digitalis* and *S. compositus* were used as replacement. *P. digitalis* was omitted from the original plant list due to high water requirements, and *S. compositus* was a last-minute substitute not originally considered.

The planting arrangement produced a plug density of 40 plugs per meter. Previous studies have shown that increasing plug density beyond 43 plugs per square meter does not increase establishment success or decrease establishment time (Snodgrass and Snodgrass, 2006). Plugs were chosen over seeds because "Green Roof Plants" by Snodgrass and Snodgrass (2006) recommended against the use of seeds, and vegetative cutting was not a viable option for the native accent species. Planting began on June 28th with Block C, but it had to be stopped due heavy rainstorms on June 29th and personal obligations on June 30th and July 1st. Blocks A and B were planted on July 2nd. The plugs were safely stored in the interim.

Establishment

Within green roof research and construction, the establishment period is the period directly after construction. During this period, the green roof receives more frequent irrigation and maintenance to ensure that the plugs, cuttings, or mats can fully take root and the groundcover species can lock the soil in place to prevent soil erosion. The establishment watering period ran from July 3rd to September 2nd. Each module was given 1 L of water at noon each day during this time period. The original plan called for 750 mL of water per plot based on watering calculations from MacIvor et al (2013). However, the summer temperatures were higher than usual, hence the amount of water was increased. All plants still alive on September 3rd were deemed to have successfully established.

Over the month of August, fifteen study plots were damaged or destroyed. Of these, seven were in Block A, three were in block B, and five were in Block C. Non-study plants growing up under the trays likely pushed and destabilized the study plots. Block A was deconstructed to fill gaps in Blocks B and C, which were used for the data collection.

Irrigation

The irrigation treatments were high irrigation, medium irrigation, and low irrigation. High irrigation modules were given 1 L of water once per week, medium irrigation modules were given 1L of water once every two weeks, and low irrigation modules were given 1L of water every 4 weeks. All irrigation was done by hand, and all irrigation was halted from November 2019 to June 2020. Irrigation treatment therefore ran from September 2nd to November 4th 2019, and from June 21st to October 12th 2020.

2.2 Data Collection and Analysis

Collection

In September of 2019 and September of 2020, vegetation cover was assessed using a 100-point pinpoint grid frame. At each point, a metal pin was lowered on a fishing wire until it struck either the substrate or vegetation. Vegetation was recorded on a 10x10 grid as a yes, with nothing being recorded for contact with substrate.

In September 2019, vegetation abundance was recorded for Block B using a 25-point pinpoint grid frame. The pin was lowered down to the top of the substrate. At each point, contacts were recorded for each species, with one contact being recorded per main stem that came in contact with the pin, even if both stems came from the same plug.

Establishment Assessment Analysis

Year one vegetation cover data was used along with vegetation abundance data to assess the relative success with which the modules in Block B were established. Using R, Block vegetation abundance data were split into the accent count dataset and the groundcover count dataset. Both datasets were checked for outliers using the default settings of the rstatix package identify_outliers function, which considers any values above (Q_3 + 1.5*IQR) or below (Q_1 – 1.5*IQR) as outliers, and any values above (Q_3 + 3*IQR) or below (Q_1 – 3*IQR) as extreme outliers. While no outliers were found in the accent count dataset, plot 11 was found to be a non-extreme outlier in the ground cover dataset. Because of this, a dataset excluding plot 11 was prepared, and all subsequent tests were run with both the outlier inclusive and outlier exclusive datasets. The Shapiro-Wilks test was used to check linear model residuals for normality. These tests found that both data sets were normally distributed. Homogeneity of variance was tested using Levene’s test. As none of the Levene’s tests were significant, the assumption of homogeneity of variance was validated. One-way ANOVAs were run on all four datasets, and the ANOVAs of the “corrected” datasets were compared to the ANOVAs of the corresponding “uncorrected” datasets to determine whether the impact of removing the single outlier had any noticeable consequence. ANOVA was chosen because the datasets met all the assumptions of an ANOVA and the variance in establishment success between the groups was more important than the actual degree of establishment success.

Survival Assessment

Cover data recorded in September 2020 was compared to cover data recorded in September 2019 in order to assess year over year survival. The dataset for each block was tested for outliers, checked for normality using Shapiro-Wilks tests, and checked for equality of variances using Levene's tests. The Levene’s tests found equal variance, but the Shapiro-Wilks tests found that the Block B observations were not normally distributed. Because of this, Kruskal-Wallis tests paired with Dunn’s Tests with Bonferroni corrections were used to analyze the data alongside ANOVAs. Independent variable ANOVAs were used rather than nested ANOVAs because accent communities were compiled and irrigation treatments were selected and administered by me, making it impossible for either to affect the other, or for time to affect either. Time, by nature of being time, is immutable. Survival analysis modeling wasn’t used because communal survival was being measured, and no community experienced complete mortality, but a survival analysis model may be useful in a longer study.

2.3 Results

Establishment Assessment

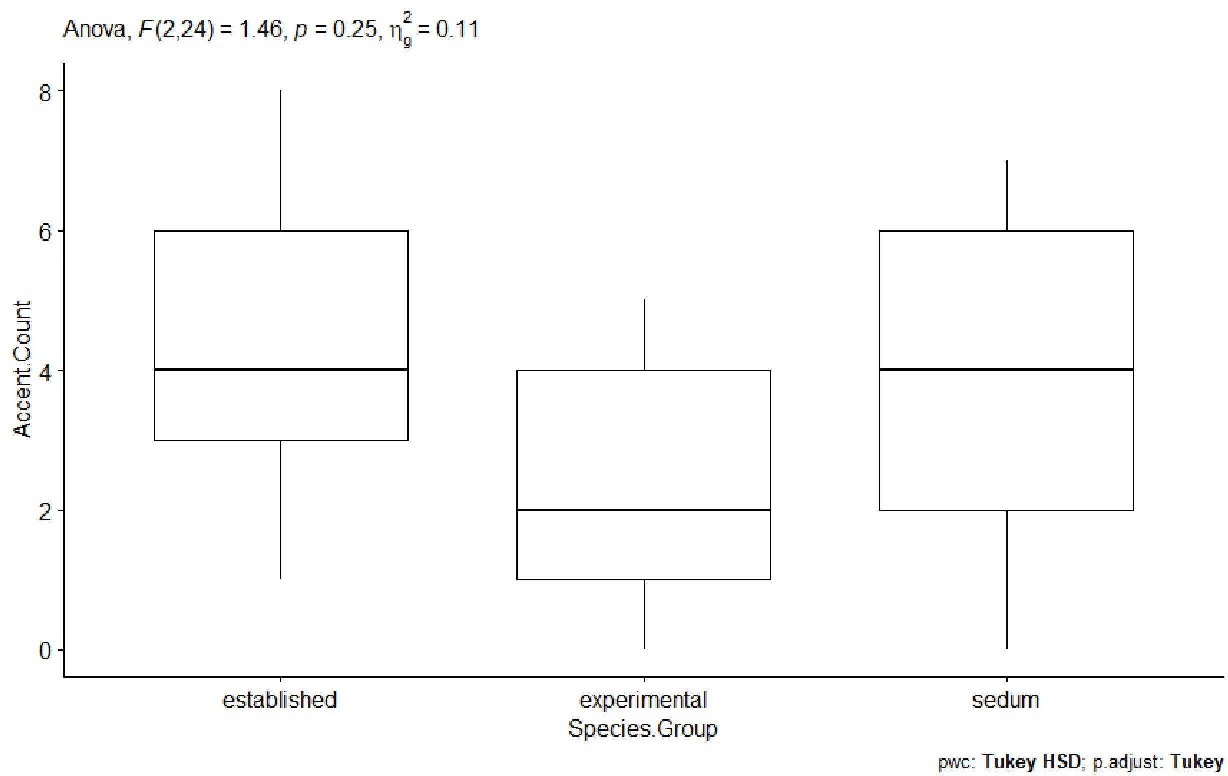


Figure 2. Boxplot and ANOVA of Post-Establishment Accent Count with Plot 11 Included

This boxplot reports the accent establishment count for all plots in Block B and the results of a one-way ANOVA. The established native accent group recorded between 1 and 8 connections on a 25-pin grid with a median value of 4. The experimental native accent group recorded between 0 and 5 connections with a median value of 2. The sedum accent group recorded between 0 and 7 connections with a median value of 4. The p-value of 0.25 suggests that we can accept the null hypothesis, and the $\eta^2{}_g$ or generalized effect size of 0.11 suggests that only a small portion of the observed variance (11%) can be attributed to the accent community.

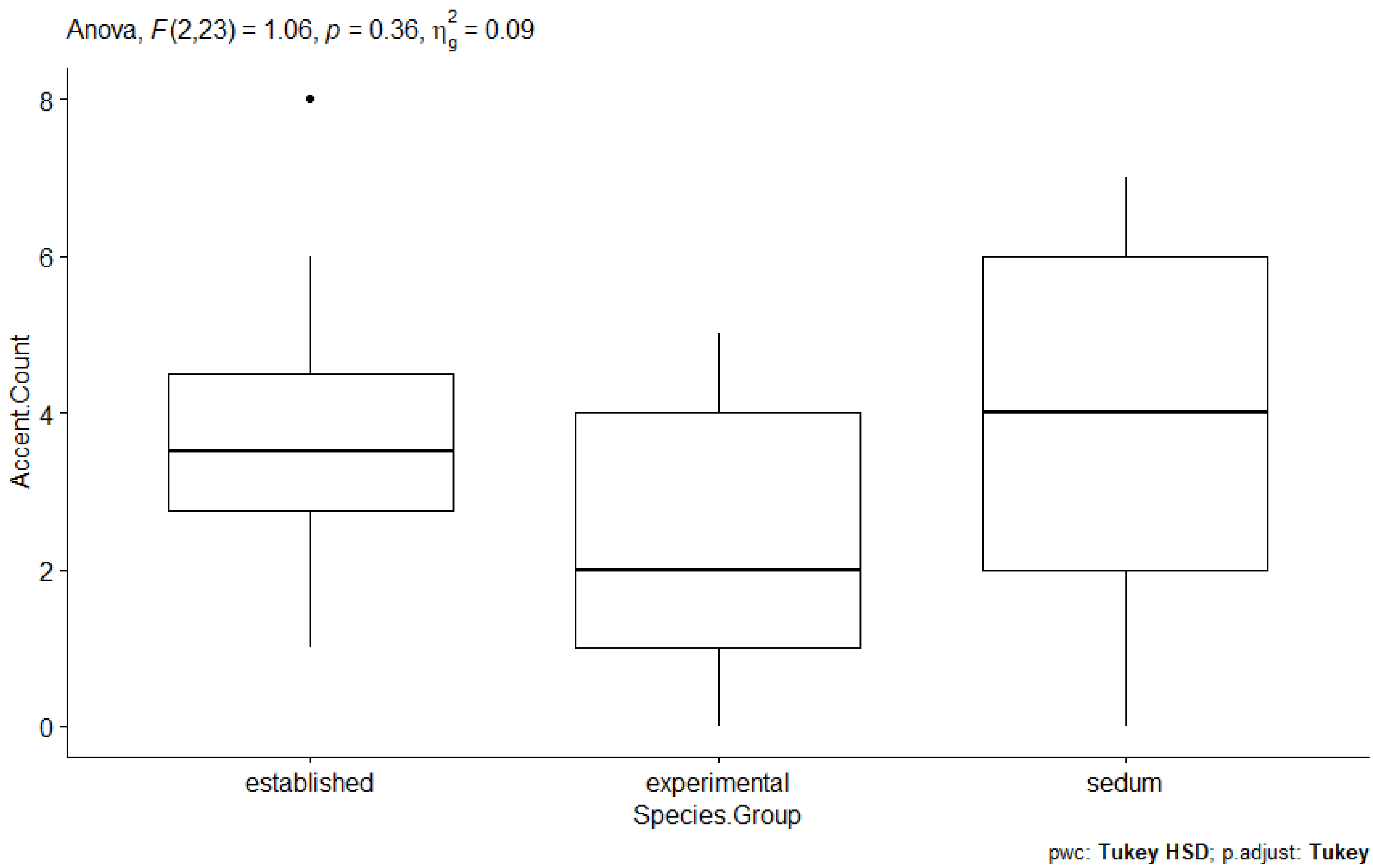


Figure 3. Boxplot and ANOVA of Post-Establishment Accent Count with Plot 11 Excluded

This boxplot reports the accent establishment count for all plots in Block B except plot 11 and the results of a one-way ANOVA. The established native accent group recorded between 1 and 8 contacts on a 25-pin grid with a median value of 3.5, though the 8 contact plot stands as a non-extreme outlier. The experimental native accent group recorded between 0 and 5 contacts with a median value of 2. The sedum accent group recorded between 0 and 7 contacts with a median value of 4. The p-value of 0.36 suggests that we can accept the null hypothesis, and the $\eta^2{}_g$ of 0.09 suggests that only a small portion of the observed variance (9%) can be attributed to the accent community.

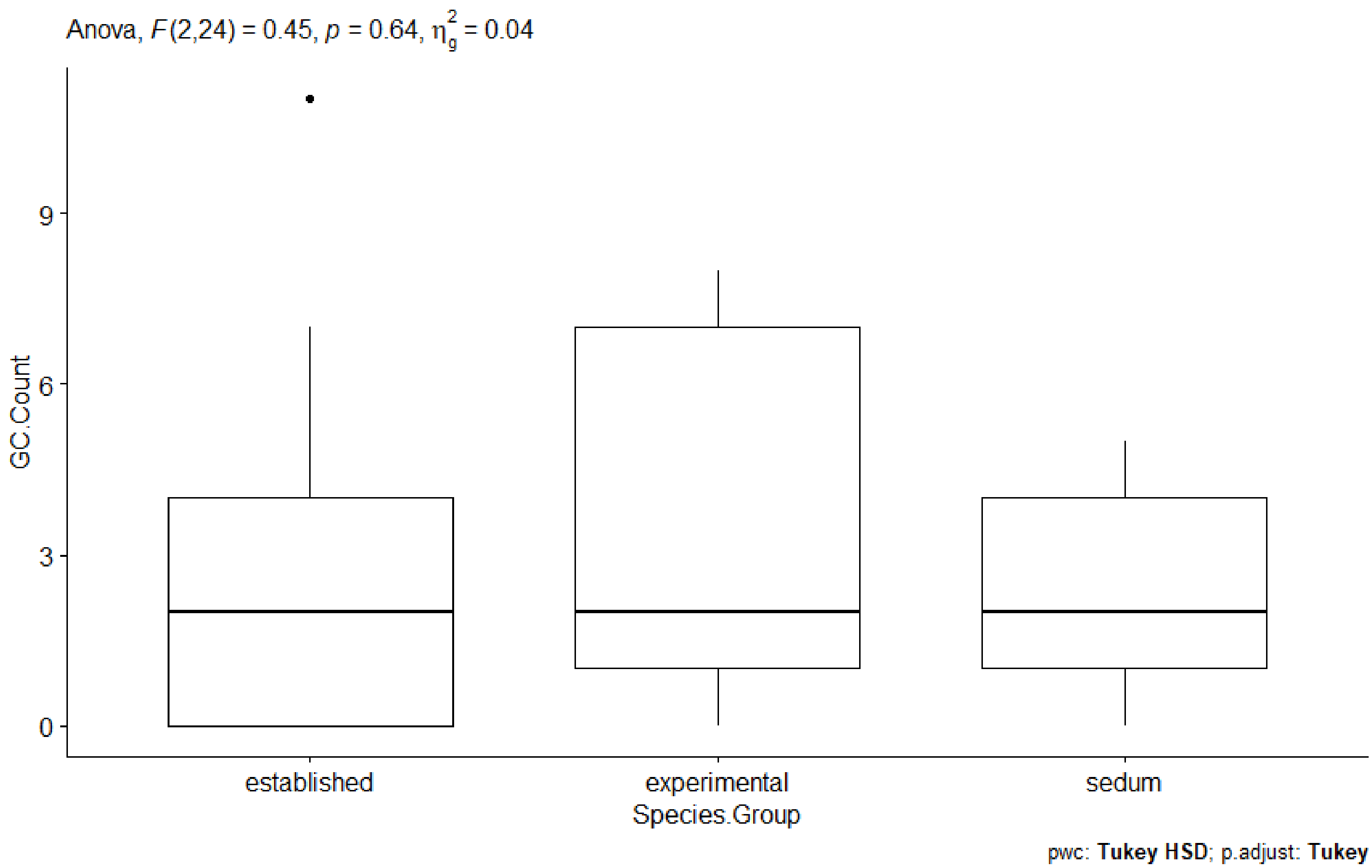


Figure 4. Boxplot and ANOVA of Post-Establishment Groundcover Count with Plot 11 Included

This boxplot reports the groundcover establishment count for all plots in Block B as well as the result of a one-way ANOVA. All three accent treatments produced a median groundcover count of 2 impacts per plot. The p-value of 0.64 suggests that we can accept the null hypothesis that the accent species present did not affect the establishment success of the sedum groundcover, and the η^2_g of 0.04 suggests that only a small portion of the observed variance (4%) can be attributed to the accent community.

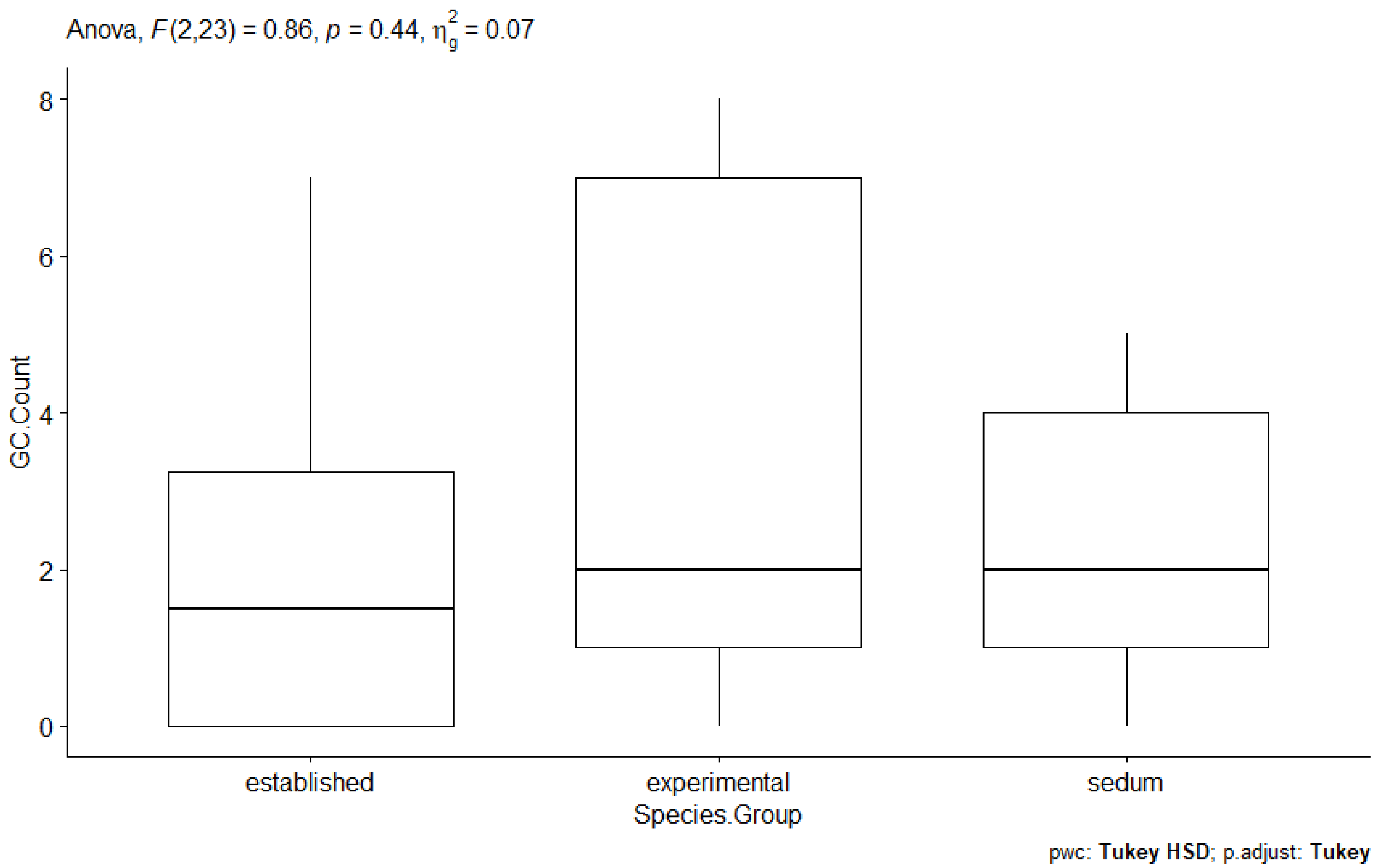


Figure 5. Boxplot and ANOVA of Post-Establishment Groundcover Count with Plot 11 Excluded

This boxplot reports the groundcover establishment count for all plots in Block B as well as the result of a one-way ANOVA. The established accent community recorded a median value of 1.5 contacts per plot. The experimental accent community a median value of 2 contacts per plot. The sedum accent community recorded a median value of 2 contacts per plot. The p-value of 0.44 suggests that we can accept the null hypothesis that the accent community present did not impact groundcover establishment, and the $\eta^2{}_g$ of 0.07 suggests that only a small portion of the observed variance (7%) can be attributed to the accent community.

Regardless of accent community, only *A. cernuum* was found to have 100% establishment according to the 3D abundance measurement, and only in within one of the two communities it was used in. Across the nine sedum plots in Block B, *S. ochroleucum* suffered a complete die-off during establishment, while *S. cauticola 'lidakense'* became established in eight out of nine plots, and *S. telephium* became established in six out of nine plots. In the nine plots planted with the established native accent species, *A. cernuum* became established in all nine plots, while *B. curtipendula* only became established in two. In the nine plots planted with the experimental native accent community, *A. cernuum* became established in eight out of nine plots, *A. canadensis*, *P. digitalis*, and *P. hirsutus* each became established in a single plot, and *B. curtipendula* and *S. compositus* both failed to establish in any plot.

Survivorship Assessment

Table 1. Block B Kruskal-Wallis Tests Analyzing Vegetation Cover

Grouping	X^2	dF	p-value
Block B All Treatments, by Year	0.77	1	0.38
Block B All Years, All Communities, by Irrigation Treatment	2.92	2	0.23
Block B All Years, All Irrigation Treatments, by Community	1.33	2	0.51
Block B 2019, All Communities, by Irrigation Treatment	1.44	2	0.49
Block B 2019, All Irrigation Treatments, by Community	1.48	2	0.48
Block B 2020, All Communities, by Irrigation Treatment	7.81	2	0.02
Block B 2020, All Irrigation Treatments, by Community	2.71	2	0.26

- The first test assessed change in cover between the 2019 and 2020 data observations without accounting for treatments.
- The second test assessed the impact of irrigation treatment without regard for year or community.
- The third test assessed the impact of irrigation without regard for year or community. The null hypothesis was not rejected.
- The fourth test assessed difference in cover between irrigation treatments in 2019, disregarding communities..
- The fifth test assessed difference in cover between communities in 2019, disregarding irrigation treatment.
- The sixth test assessed difference in cover between irrigation treatments in 2020, disregarding communities. The null hypothesis that irrigation treatment did not affect vegetation cover observations from 2020 was rejected. This is explored further in table 6.

- The seventh test assessed difference in cover between communities in 2020, disregarding irrigation treatment.

Table 2. Dunn Test with Bonferroni Correction Block B All Years, All Communities, By Irrigation Treatment

Comparison	Z	P. unadjusted	P. adjusted
High-Low	1.65	0.098	0.295
High-Medium	1.20	0.230	0.689
Low-Medium	-0.452	0.651	1.000

Table 2 reports the results of a Dunn's post-hoc test with a Bonferroni correction applied to the second Kruskal-Wallis test from Table 1, difference in cover between irrigation treatments without regard for year of observation or community observed. When compared without consideration for accent community observed or year of observation, no two irrigation treatments showed enough difference in vegetation cover to reject the null hypothesis that irrigation treatment did not affect vegetation cover.

Table 3. Dunn Test with Bonferroni Correction Block B All Years, All Irrigation Treatments, by Community

Comparison	Z	P. unadjusted	P. adjusted
Established-Experimental	-0.101	0.92	1.00
Established-Sedum	-1.05	0.295	0.344
Experimental-Sedum	-0.946	0.344	1.00

Table 3 reports the results of a Dunn's post-hoc test with a Bonferroni correction applied to the third Kruskal-Wallis test from Table 1, difference in cover between accent communities without regard for year of observation or irrigation treatment. When compared without consideration for irrigation treatment observed or year of observation, no two accent communities showed enough difference in vegetation cover to reject the null hypothesis that accent community did not affect vegetation cover.

Table 4. Dunn Test with Bonferroni Correction Block B 2019, All Communities, by Irrigation Treatment

Comparison	Z	P. unadjusted	P. adjusted
High-Low	0.388	0.698	1.000
High-Medium	-0.790	0.429	1.000
Low-Medium	-1.18	0.239	0.716

Table 4 reports the results of a Dunn's post-hoc test with a Bonferroni correction applied to the fourth Kruskal-Wallis test from Table 1, difference in vegetation cover between irrigation treatments without regard for accent community, based on 2019 observations. When compared without consideration for accent community observed, no two irrigation treatments showed enough difference in vegetation cover observed in 2019 to reject the null hypothesis that irrigation treatment did not affect vegetation cover observed in 2019.

Table 5. Dunn Test with Bonferroni Correction Block B 2019, All Irrigation Treatments, by Community

Comparison	Z	P. unadjusted	P. adjusted
Established-Experimental	1.09	0.276	0.829
Established-Sedum	0.075	0.941	1.000
Experimental-Sedum	-1.01	0.311	0.932

Table 5 reports the results of a Dunn's post-hoc test with a Bonferroni correction applied to the fifth Kruskal-Wallis test from Table 1, difference in vegetation cover observed in 2019 between accent communities, without regard for irrigation treatment. When compared without consideration for irrigation treatment, no two accent communities showed enough difference in vegetation cover observed in 2019 to reject the null hypothesis that accent community did not affect vegetation cover observed in 2019.

Table 6. Dunn Test with Bonferroni Correction Block B 2020, All Communities, by Irrigation Treatment

Comparison	Z	P. unadjusted	P. adjusted
High-Low	1.94	0.053	0.158
High-Medium	2.71	0.007	0.02
Low-Medium	0.775	0.438	1.000

Table 6 reports the results of a Dunn's post-hoc test with a Bonferroni correction applied to the sixth Kruskal-Wallis test from Table 1, difference in vegetation cover observed in 2020 between irrigation treatments, without regard for accent communities. As stated above, the Kruskal-Wallis test was able to reject the null hypothesis, and determine that irrigation treatment did in fact have an impact on vegetation cover. According to the results of the Dunn's test, the null hypothesis could only be rejected for the high irrigation versus medium irrigation comparison.

Table 7. Dunn Test with Bonferroni Correction Block B 2020, All Irrigation Treatments, by Community

Comparison	Z	P. unadjusted	P. adjusted
Established-Experimental	-1.28	0.200	0.599
Established-Sedum	-1.54	0.125	0.374
Experimental-Sedum	-0.253	0.800	1.000

Table 7 reports the results of a Dunn's post-hoc test with Bonferroni correction applied to the seventh Kruskal-Wallis from Table 1, difference in vegetation cover, as observed in 2020, between accent communities, without regard for irrigation treatment. When compared without consideration for irrigation treatment, no two accent communities showed enough difference in vegetation cover observed in 2020 to reject the null hypothesis that accent community did not affect vegetation cover observed in 2020.

Table 8. Results of a Three-Way ANOVA Analyzing the Impact of Community and Irrigation Level on Year One Survivorship Based on Percent Vegetation Cover in Block B

Effect	**DFn**	**DFd**	**F**	**p**	**ges**
Irrigation	2	36	1.31	0.28	0.07
Community	2	36	1.13	0.34	0.06
Year	1	36	0.89	0.35	0.02
Irrigation:Community	4	36	2.21	0.09	0.20
Irrigation:Year	2	36	2.96	0.07	0.14
Community:Year	2	36	1.70	0.20	0.09
Irrigation:Community:Year	4	36	1.13	0.36	0.11

Table 8 shows the results of a three-way ANOVA comparing the impact of irrigation, the accent community, and the year on percent plant coverage in Block B. In this context, the effect of the year encapsulates all the environmental conditions between September 2019 and September 2020 that are beyond any control, but which should have remained relatively consistent across all plots. While no factor or combination of factors produced a p-value small enough to justify rejecting the null hypothesis, the two-way interactions between irrigation and community and between irrigation and the year produced the smallest p-values, and the two-way interaction between the irrigation treatment and the accent community explained more of the variance than three-way interaction accounting for time. Analysis of the interactions between community treatment and irrigation treatment produced a p value less than 0.2, as did the analysis of irrigation treatment impact over time, so while the null hypotheses for these interactions were not rejected, all three sets of interactions should be considered in the design of a future study.

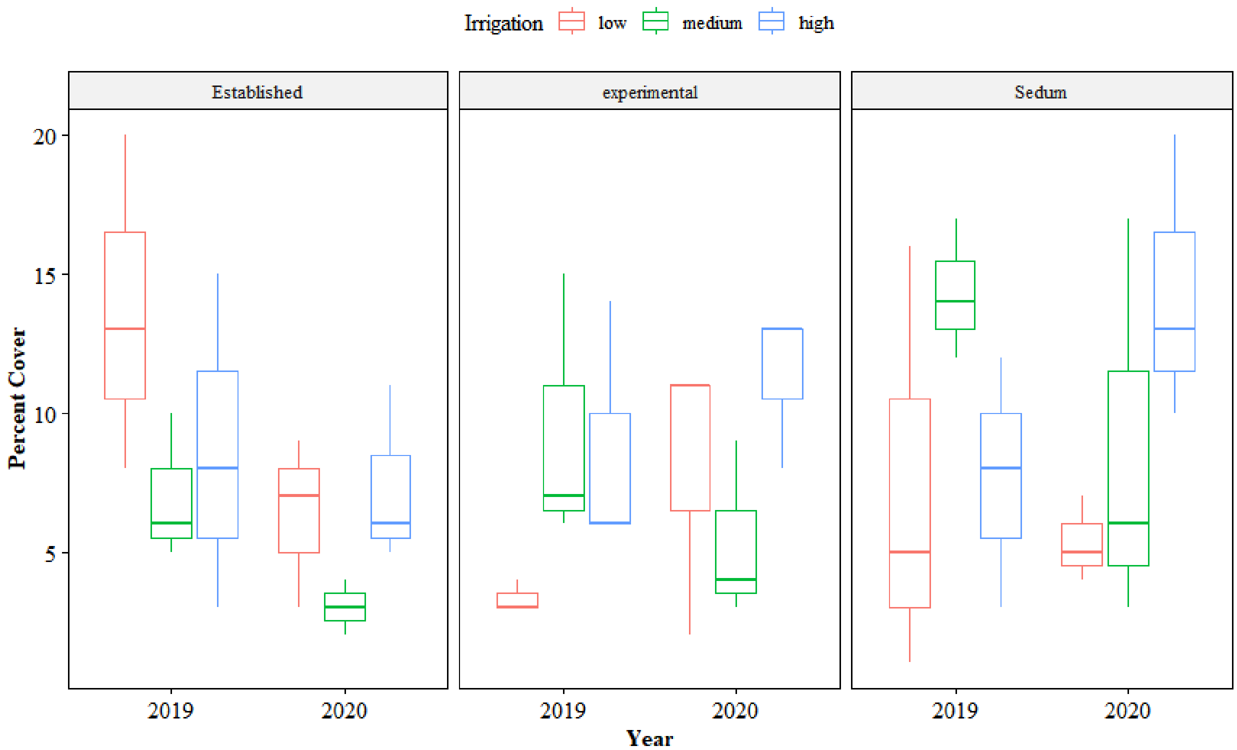


Figure 6. Boxplot of Percent Vegetation Cover Observations for Block B from 2019 and 2020

Figure 6 shows a set of grouped boxplots depicting the change in cover for each community in Block B when separated based on irrigation levels. In the plots planted with established native species, median and maximum cover decreased between observations at all irrigation levels, whereas experimental native saw an increase in median cover at high and low irrigation levels, but a decrease in median, minimum, and maximum cover at medium irrigation, and in the sedum community plots, range decreased while median cover stayed the same at low irrigation, median and minimum cover decreased at medium irrigation, and while only high and medium irrigation sedum plots managed to maintain cover greater than 15%, no plot in Block B suffered complete loss of cover between 2019 and 2020.

Table 9. Block C Kruskal-Wallis Tests

Grouping	X^2	dF	p-value
Block C All Treatments, by Year	5.04	1	0.02
Block C All Years, All Communities, by Irrigation Treatment	3.72	2	0.16
Block C All Years, All Irrigation Treatments by Community	1.33	2	0.51
Block C 2019, All Communities, by Irrigation Treatment	1.92	2	0.39
Block C 2019, All Irrigation Treatments, by Community	6.71	2	0.03
Block C 2020, All Communities, by Irrigation Treatment	3.56	2	0.17
Block C 2020, All Irrigation Treatments, by Community	0.55	2	0.76

Table 9 reports the results of Kruskal-Wallis tests applied to observations of block C. The first test assessed change in cover between the 2019 and 2020 data observations without accounting for treatments. The null hypothesis was rejected, suggesting that across all treatments for block C, there was a statistically significant change in vegetation cover between 2019 and 2020. The second test assessed the impact of irrigation treatment without regard for year or community. The null hypothesis could not be rejected. The third test assessed the impact of irrigation without regard for year or community. The null hypothesis could not be rejected. The fourth test assessed difference in cover between irrigation treatments in 2019, disregarding communities. The null hypothesis could not be rejected. The fifth test assessed difference in cover between communities in 2019, disregarding irrigation treatment. The null hypothesis was rejected, meaning that without accounting for the impact of irrigation treatment, there was a statistically significant difference in vegetation cover between the accent communities at the time of the 2019 data collection. The sixth test assessed difference in cover between irrigation treatments in 2020, disregarding communities. The null hypothesis could not be rejected. The seventh test assessed difference in cover between communities in 2020, disregarding irrigation treatment. The null hypothesis could not be rejected.

Table 10. Dunn Test with Bonferroni Correction Block C All Years, All Communities, By Irrigation Treatment

Comparison	Z	P. unadjusted	P. adjusted
High-Low	0.462	0.644	1.000
High-Medium	-1.43	0.154	0.462
Low-Medium	-1.82	0.069	0.206

Table 10 reports the results of a Dunn’s post-hoc test with a Bonferroni correction applied to the second Kruskal-Wallis test from Table 9, difference in cover between irrigation treatments without regard for year of observation or community observed. When compared without consideration for accent community observed or year of observation, no two irrigation treatments showed enough difference in vegetation cover to reject the null hypothesis that irrigation treatment did not affect vegetation cover.

Table 11. Dunn Test with Bonferroni Correction Block C All Years, All Irrigation Treatments, by Community

Comparison	Z	P. unadjusted	P. adjusted
Established-Experimental	1.02	0.309	0.927
Established-Sedum	-0.865	0.387	1.000
Experimental-Sedum	-1.89	0.060	0.179

Table 11 reports the results of a Dunn's post-hoc test with a Bonferroni correction applied to the third Kruskal-Wallis test from Table 9, difference in cover between accent communities without regard for year of observation or irrigation treatment. When compared without consideration for irrigation treatment observed or year of observation, no two accent communities showed enough difference in vegetation cover to reject the null hypothesis that accent community did not affect vegetation cover.

Table 12. Dunn Test with Bonferroni Correction Block C 2019, All Communities, by Irrigation Treatment

Comparison	Z	P. unadjusted	P. adjusted
High-Low	0.866	0.386	1.000
High-Medium	-0.526	0.599	1.000
Low-Medium	-1.38	0.169	0.506

Table 12 reports the results of a Dunn's post-hoc test with a Bonferroni correction applied to the fourth Kruskal-Wallis test from Table 9, difference in vegetation cover between irrigation treatments without regard for accent community, based on 2019 observations. When compared without consideration for accent community observed, no two irrigation treatments showed enough difference in vegetation cover observed in 2019 to reject the null hypothesis that irrigation treatment did not affect vegetation cover observed in 2019.

Table 13. Dunn Test with Bonferroni Correction Block C 2019, All Irrigation Treatments, by Community

Comparison	Z	P. unadjusted	P. adjusted
Established-Experimental	1.68	0.093	0.280
Established-Sedum	-0.912	0.362	1.000
Experimental-Sedum	-2.65	0.010	0.031

Table 13 reports the results of a Dunn's post-hoc test with a Bonferroni correction applied to the fifth Kruskal-Wallis test from table 9, difference in vegetation cover observed in 2019 between accent communities, without regard for irrigation treatment. As stated above, the Kruskal-Wallis test was able to reject the null hypothesis and determine that there was statistically significant variation in vegetation between the accent communities at the time of the 2019 data collection. Table 13 explores which comparisons show that variation. When the established native accent community is compared to the experimental native accent community, the null hypothesis cannot be rejected. The same is true of the comparison between the established native accent community and the sedum accent community. Only the comparison between the sedum accent community and the experimental native accent community was able to reject the null hypothesis.

Table 14. Dunn Test with Bonferroni Correction Block C 2020, All Communities, by Irrigation Treatment

Comparison	Z	P. unadjusted	P. adjusted
High-Low	0.004	0.997	1.000
High-Medium	-1.67	0.096	0.287
Low-Medium	-1.56	0.118	0.355

Table 14 reports the results of a Dunn’s post-hoc test with a Bonferroni correction applied to the sixth Kruskal-Wallis test from table 9, difference in vegetation cover observed in 2020 between irrigation treatments, without regard for accent communities. The original Kruskal-Wallis test was unable to reject the null hypothesis that at the time of the 2020 data collection, there was not a statistically significant variation in vegetation cover between the irrigation treatment groups, and the pairwise comparisons of the irrigation treatments in table 14 each also failed to reject the null hypothesis.

Table 15. Dunn Test with Bonferroni Correction Block C 2020, All Irrigation Treatments, by Community

Comparison	Z	P. unadjusted	P. adjusted
Established-Experimental	-0.669	0.504	1.0
Established-Sedum	-0.614	0.539	1.0
Experimental-Sedum	0.075	0.941	1.0

Table 15 reports the results of a Dunn's post-hoc test with a Bonferroni correction applied to the seventh Kruskal-Wallis test from table 9, difference in vegetation cover observed in 2020 between accent communities, without regard for irrigation treatments. The original Kruskal-Wallis test was unable to reject the null hypothesis that at the time of the 2020 data collection, there was not a statistically significant variation in vegetation cover between the accent communities, and the pairwise comparisons of the accent communities in table 15 each also failed to reject the null hypothesis.

Table 16. Results of a Three-Way ANOVA Analyzing the Impact of Community and Irrigation Level on Year One Survivorship Based on Percent Vegetation Cover in Block C

Effect	**DFn**	**DFd**	**F**	**p**	**ges**
Irrigation	2	33	3.142	0.056	0.160
Community	2	33	3.255	0.051	0.165
Year	1	33	6.325	0.017	0.161
Irrigation:Community	4	33	0.777	0.548	0.086
Irrigation:Year	2	33	0.457	0.637	0.027
Community:Year	2	33	2.082	0.141	0.112
Irrigation:Community:Year	4	33	0.433	0.784	0.050

Table 16 shows the results of a three-way ANOVA comparing the impact of irrigation, the accent community, and the year on percent plant coverage in Block C. In this context, the effect of the year encapsulates all the environmental conditions between September 2019 and September 2020 that were beyond any control, but which should have remained relatively consistent across all plots. Unlike Block B, Block C did show a statistically significant decrease in vegetation cover, suggesting that some external factor impaired survivorship in Block C. The null hypothesis that no community would be disproportionately affected by irrigation levels could not be rejected. The null hypothesis that irrigation level would not affect plant cover over time similarly could not be rejected. While the null hypothesis that survivorship over time would not vary between the three plant communities could not be rejected, the impact of community over time analysis produced a p value less than 0.2, reinforcing the assertion from Block B that survivorship over time between the different communities should be a major consideration in any future studies.

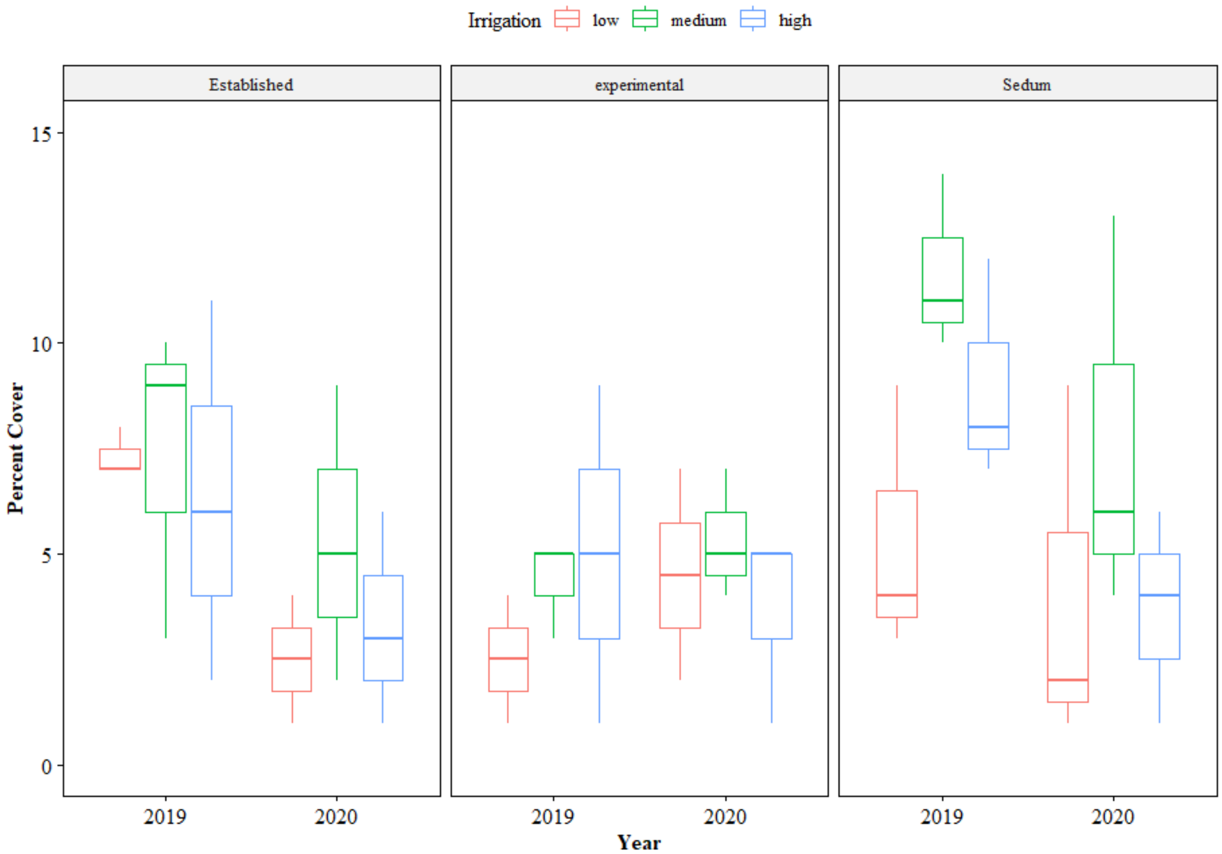


Figure 7. Boxplot of Percent Vegetation Cover Observations for Block C from 2019 and 2020

Figure 7 shows a set of grouped boxplots depicting the change in cover for each accent community in Block C when separated based on irrigation treatment. As with Block B, no plot suffered complete loss of vegetation. Both the established natives community and the sedum community experienced a decrease in median percent cover across all irrigation levels. The experimental natives community maintained median cover at median and high irrigation levels, and cover increased for the low irrigation treatment.

2.5 Discussion

Addressing Q1: How effectively do green roofs with native accent species establish themselves relative to green roofs planted with exclusively sedum species?

The results for the accent and groundcover communities, shown in figures 2, 3, 4, and 5, suggested that overall establishment success did not significantly vary between accent communities. Based on my proof-of-concept study, a longer-term research project can – and should – use these species to test how more variables might influence medium term outcomes, e.g. persistence and ecological impact. However, my shorter-term study did show that some species may be more persistent than others and might yield more useful outcomes in a green roof design. Figures 3 and 4 indicate that accent communities had little or no impact on groundcover species. This is in in keeping with prevalent thinking in green roof design that groundcover species are primarily responsible for green roof success, with accent species mainly playing an aesthetic role (Luckett, 2009; Snodgrass 2006). Figures 1 and 2 indicate that on a community level, establishment success was not impacted by the community that was present. At a base level, the argument can thus be made that accent species make up did not impact green roof establishment. For a more in-depth analysis, we need to focus on the comparative success of the individual accent species.

At the end of the establishment period, three accent species had apparently failed completely: *S. ochroleucum* failed in the sedum accent community, and *B. curtipendula* and *S. compositus* failed in the experimental native accent community. *S. cauticola 'lidakense'* and *S. telephium* from the sedum accent community outsurvived all other species except *A. cernuum*, which had total survival in the established native accent species community, and near total survival in the experimental native accent community. It is important to note, however, that this qualitative survival metric was not directly observed, but was extrapolated from the data reported in figures 1 and 2. The 25 grid points used to collect these data were spaced 100mm apart orthogonally and 141mm apart diagonally. The large spacing between the points may have allowed for some species to survive while not being detected, an issue which could be solved by incorporating a qualitative check for each species used in future studies.

A. cernuum is the only native accent species that reliably established itself. This result is corroborated by Getter et al (2009), who found that *A. cernuum* is able to perform comparably to sedum species on green roofs, and by Vandegrift et al (2019), who found that of the 22 species planted, *A. cernuum* had the highest survival rate and highest absolute cover at soil depths of 10cm and 20cm. *A. cernuum* should therefore be considered a top candidate for any green roof design intending to use native species.

S. compositus from the experimental accent community and *S. ochroleucum* from the sedum accent community both failed completely. *S. compositus* was a last minute replacement for *Sporobolus cryptandrus*, which, according to Changgui et al(1993), takes in most of its water from the 0-30cm soil zone, and which is able to grow both in mountainous regions and in USDA hardiness zone 5, where the field site was located. *S. cryptandrus* was originally chosen for its similarity to *Sporobolus heterolepis*, a species known to be viable on green roofs according to Snodgrass (2006). According to profiles of the three species in the Fire Effects Information System (FEIS) of USDA Forest Service, the three Sporobolus species share many similarities, but of the three, only *S. compositus* is not found in montane habitats. Sylvie et al (2022) found that montane habitats tended to yield native species suitable for green roofs. This alongside the failure of *S. compositus* despite its tolerance for the soil and drought conditions on green roofs may suggest that the ability to withstand montane conditions should be a parameter for the selection of native species to be used in green roof experiments.

A. canadensis, *P. digitalis*, *P. hirsutus*, and *B. curtipendula* each survived in at least one module. *P. digitalis* was expected to have survival issues. The species was originally rejected for this study due to its drought vulnerability before it becomes established and was only added as a replacement for more suitable species that failed in the nursery. *A. canadensis*, *P. hirsutus*, and *B.* curtipendula, however, were each expected to have higher survival rates. *A. canadensis* and *B. curtipendula* were species recommended by “Green Roof Plants” (Snodgrass 2006). Natvik (2012) included surveys of native green roof case studies in southern Ontario, all of which were unirrigated. Of the eleven green roofs surveyed, *A. canadensis* was found growing successfully on eight, *B. curtipendula* was found on five, and *P. hirsutus* was found on nine. Of particular interest is the Activa Sportplex green roof, which is located less than 10 kilometers from my study site and has the same soil depth as the test plots in this study. A study conducted in London, ON, found 100% two-year survival of *A. canadesis* on green roofs when seeded as a part of a mixture including *S. spurium* and *Sporobolus heterolepis* (Tran et al, 2019). One possible explanation for the high failure rate of *A. canadensis* and *B. curtipendula* may have been a failure in establishment design. One liter of water per 0.25 m^2 plot per day may have been insufficient for the species chosen. This watering regimen was selected based on previous work that used known viable green roof species, and may have been incompatible with the species communities chosen. Future studies should experiment with higher and lower establishment watering regimens to assess their impact on establishment. A secondary explanation for the failure of *A. canadensis* may have been omission. As part of the surveying method, only contacts with visibly living tissue were reported. At least one contact with an apparently dead specimen of *A. canadensis* was noted, but not included in the quantitative data. *A. canadensis*, however, is a wildflower that blooms from June through August. Had the survey been conducted during the plant's blooming season, higher survival may have been reported. This could be addressed in future studies by either not discounting contacts with apparently dead specimen post-blooming season, or by conducting measurements multiple times during the establishment phase.

Addressing Q2&3: How do overwintering and reduced irrigation affect the survival of green roofs populated with native accent species compared to green roofs populated with sedum accents?

The second and third questions need to be addressed together. In addressing these questions, three points should be made clear. The first issue is that total vegetation cover will be used as a proxy measurement for groundcover community survival. As mentioned in the introduction, groundcover species are the backbone of successful green roofs. They stabilize the substrate and provide the vast majority of vegetation cover. "Green Roof Plants" even suggests that accent species die off and re-sowing may be part of long-term maintenance of a healthy green roof. The second point is that the Kruskal-Wallis tests with Bonferroni corrections were in agreement with the one-way components of the three-way ANOVAs. The third point is that the data and ANOVA results for blocks B and C conflict, and that conflict becomes stronger when multiple α-levels are considered. Even at α-level 0.20, it cannot be concluded that either the community or irrigation treatment had a significant impact on vegetation cover of plots in Block B, and vegetation cover of plots in Block B did not significantly change between September 2019 and September 2020. Block C, however, is another matter entirely. At α-level 0.20, both the community and irrigation treatments had a significant impact on vegetation cover, and there was significant change in vegetation between September 2019 and September 2020. All three effects remain significant at α-level 0.10, and the change in vegetation over time is only insignificant if the α-is set to 0.01. Strangely, however, when irrigation treatment is taken into account with community treatment, the result is insignificant at any α-level, especially when looked at over time. The impact of irrigation by itself over time on Block C is also insignificant. The impact of community over time is significant at α-level 0.20, but not at lower α-levels. This seems to suggest some unobserved effect caused plots in Block C to lose cover over time, but the accent communities were not equally affected. One possible explanation for the discrepancy between the two test blocks is shading. Getter et al (2009) explored the effect variable shading can have on green roof plant communities. At the field site where the two test blocks were set up, there is a tree line that cast shade on the blocks in the evenings. There is, however, a break in this tree line large enough to have allowed one block more sunlight exposure than the other at certain times of the growing season. While not originally a consideration of the study, this is similar to the impact shading from neighboring tall buildings could have on the green roofs, suggesting that shading might be an additional impact to consider in the long-term study.

MacIvor et al (2013) found that irrigation was not a serious driver of plant cover or biomass in traditional green roofs. The results of the ANOVAs support this, though the results of the sixth Kruskal-Wallis test described in table 1, "Block B 2020, All Communities, By Irrigation Treatment," combined with the accompanying Dunn Bonferroni test detailed in Table 6 suggest that there was some degree of discrepancy between the high irrigation group and the medium irrigation group. The caveat is that my results may be less comparable because of the unavoidable disruption to the irrigation treatment caused by the original Covid-19 lockdowns, i.e. irrigation treatments could not be started until July 2020. This meant that no irrigation occurred during the first two months of the growing season, and resulted in the low irrigation treatment plots only being watered three times over the course of the second growing season. The lack of significant loss of cover year over year despite this disruption suggests that these green roof designs in this climate region do not strictly require supplemental irrigation at all. This observation lines up with the conclusion made in MacIvor et al (2013) based on a study conducted in Toronto, Ontario, that while supplemental irrigation is important for biodiversity, biomass and vegetation cover are not significantly impacted when supplemental irrigation is disrupted. This also makes logical sense because the groundcover community is the primary contributor to vegetation cover, and all three groundcover species were *Sedums* commonly used in green roofs. Accent community treatment also did not appear to have a significant impact on vegetation cover. At α-level 0.10, Block C appeared to show a difference in cover between the three accent communities at specific times, but this difference disappears when you look beyond individual snapshots and compare relative change over time, suggesting that if there was a difference, it was primarily during the establishment phase, and post-establishment survival is relatively consistent between the three groups. This discrepancy is not present in Block B.

The next question is why neither irrigation nor accent community impacted vegetation cover. The assumption was made that because the groundcover species used were three of the most reliable *Sedum* species used in green roofs, any impact caused by accent communities would have been negative, caused by competition between the accent and groundcover communities. However, rather than competing, the *Sedum* species may have helped the accent species withstand the period when irrigation was disrupted. Butler et al (2011) found that *sedum* species reduced peak soil temperature, which would have minimized evaporation of water in the upper portion of the substrate, potentially stretching the impact of snow melt and limited rainfall until irrigation could be resumed. This could be tested in a long-term study through regular substrate moisture measurements. Tran et al (2019) found that green roofs planted with a mixture of *A. canadensis*, *Sporobolus heterolepis*, and *S. spurium* were more successful than those planted with any of the three species in monoculture. Matsuoka et al (2019) also found that certain *Sedum* species improved the performance of nectar-producing plants. Their study could be germane in the longer term, as the intent of this project is to eventually produce a green roof design that will facilitate the movement of pollinator communities.

Beyond the three major questions, the primary goal of this project was to inform the design of a long-term, large-scale test of a hybrid green roof design. The final section of this thesis will therefore address limitations in the proof-of-concept, identify additional questions for the long-term study to address, and propose a potential design for the long-term study. The proof-of-concept study had two major points of failure. The first point of failure was the loss of Block A, which occurred as a result of plants growing up under the trays, causing them to destabilize and crack. If the long-term study is set up as an actual green roof, this problem should resolve itself. The second point of failure was the disruption of the irrigation treatment that occurred when the University of Waterloo ordered all fieldwork to be suspended during the first Covid-19 lockdown. As a result of this, an irrigation treatment that could have started as early as late April instead started in late June. Such a disruption is inherently a risk if irrigation is conducted manually. In the long-term study, this can be avoided by equipping the test green roofs with a built-in irrigation system that activates either automatically or by remote control.

The proof-of-concept study only focused on the relative viability of the three proposed plant communities, however there is more to a green roof than plant survival. Previous work has shown that certain groupings of plant forms can improve ecosystem services, but not all plant form groupings do this to the same degree (Lundholm et al, 2010). Therefore, the long-term study must assess the impact on UHI effect mitigation, stormwater run-off and storm surge mitigation, and reduction in energy requirements. UHI mitigation can be measured using surface temperature at various times of day and times of year as a proxy metric. Reduction in energy use can be measured using the difference in temperature between the surface and the bottom of the substrate layer as an assessment of insulative capacity. Stormwater run-off and storm surge mitigation can be measured by incorporating a flow rate meter into the drainage system and recording the total stormwater capture, peak flow rate, and time of peak flow rate during each storm event. In addition to the three main ecosystem services, two other environmental impacts should be assessed. The first is impact on stormwater run-off water quality, which can be assessed by chemical analysis of runoff samples collected from each system. The second and more important is the impact of the green roofs on arthropod communities. This would require a arthropod community survey of each site to be conducted the year before the construction of the green roofs, as well as every year the study continues for.

This leads to a reflection on a design for a long-term-study. As the green roof design is intended to be used across Southern Ontario, study sites should be established in multiple cities. Waterloo-Kitchener and Toronto are the primary cities that should be selected (KW because that's where the University of Waterloo is located and Toronto because of Toronto's green roof bylaw), but other cities should be considered. Each city should have at least two sites, one closer to the urban center and one closer to the outskirts. This will provide an idea of how urban density affects the ability to connect to arthropod communities. Each site should consist of nine separate green roof systems, each representing one of the nine community-irrigation treatment combinations. Each system should be at least 50 m^2 for a total of at least 450 m^2, slightly larger than the smallest green roof required by the Toronto green roof by-law. The green roofs should be constructed in place, not using modules, and contain 150 mm of growing medium. As mentioned above, each green roof system should have a separate built-in irrigation system that either activates automatically or can be activated remotely. Accent communities should be seeded as plugs. Sedum groundcover species should be seeded as plugs, then overseeded with cuttings to fill empty gaps. The groundcover species used should include *S. kamtschaticum*, *S. album*, *S. spurium*, and *S. sexangular*. The established native treatment should include *A. curnuum*, *A. canadensis*, and *B. curtipendula*. The experimental native treatment should include all three established native species, as well as *P. hirsutus* and other species depending on availability.

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Appendix A. Ground Cover Species Table

Family	Genus	Number of Used Species
Aizoaceae	Delosperma	12
Asphodelaceae	Bulpine	1
Asteraceae	Achillea	2
Asteraceae	Antennaria	2
Asteraceae	Artemisia	1
Asteraceae	Eriophyllum	1
Asteraceae	Hieracium	6
Brassicaceae	Alyssum	2
Brassicaceae	Aurinia	1
Caryophyllaceae	Cerastium	1
Caryophyllaceae	Petrorhagia	1
Caryophyllaceae	Saponaria	1
Caryophyllaceae	Silene	4
Crassulaceae	Crassula	1
Crassulaceae	Orostachys	2
Crassulaceae	Sedum	37
Illecebraceae	Herniaria	1
Lamiaceae	Malephora	2
Lamiaceae	Thymus	3
Papilionaceae	Lotus	1
Poaceae	Buchloe	1
Polemoniaceae	Phlox	1
Rosaceae	Alchemilla	1
Rosaceae	Potentilla	3
Scrophulariaceae	Veronica	4

This table was compiled from the list of plants found in the book "Green Roof Plants" by Edmund Snodgrass and Lucie Snodgrass (2006). Please see said book for a complete list of species as well as the requirements for each species used.

Appendix B. Accent Species Table

Family	Genus	Number of Used Species
Aizoaceae	Delosperma	2
Aizoaceae	Ruschia	1
Alliaceae	Allium	7
Alliaceae	Triteleia	1
Alliaceae	Tulbaghia	1
Asteraceae	Anacyclus	1
Asteraceae	Anthemis	1
Asteraceae	Artemisia	1
Asteraceae	Aster	3
Asteraceae	Chrysopsis	1
Asteraceae	Erigeron	3
Asteraceae	Othonna	1
Asteraceae	Santolina	1
Asteraceae	Townsendia	1
Boraginaceae	Echium	2
Brassicaceae	Aethionema	1
Cactaceae	Opuntia	1
Campanulaceae	Campanula	1
Caryophyllaceae	Arenaria	1
Caryophyllaceae	Dianthus	6
Caryophyllaceae	Lychnis	1
Crassulaceae	Jovibarba	2
Crassulaceae	Kalanchoe	1
Crassulaceae	Orostachys	1
Crassulaceae	Rosularia	2
Crassulaceae	Sedum	22
Crassulaceae	Sempervivum	6
Cyperaceae	Carex	2
Dipsacaceae	Scabiosa	1
Ericaceae	Calluna	1
Euphorbiaceae	Euphorbia	1
Hyacinthaceae	Scilla	1
Hydrophyllaceae	Phacelia	1
Iridaceae	Iris	2
Lamiaceae	Agastache	3
Lamiaceae	Dracocephalum	1
Lamiaceae	Lavandula	1
Lamiaceae	Origanum	1
Lamiaceae	Prunella	1

Lamiaceae	Salvia	4
Lamiaceae	Scutellaria	2
Linaceae	Linum	2
Onagraceae	Oenothera	3
Papaveraceae	Papaver	1
Papilionaceae	Anthyllis	1
Plumbaginaceae	Armeria	2
Plumbaginaceae	Goniolimon	1
Poaceae	Bouteloua	2
Poaceae	Deschampsia	1
Poaceae	Festuca	1
Poaceae	Koeleria	2
Poaceae	Marrubium	1
Poaceae	Nassella	1
Poaceae	Poa	2
Poaceae	Seslaria	2
Poaceae	Sporobolus	1
Polygonaceae	Eriogonum	1
Portulacaceae	Talinum	4
Ranunculaceae	Aquilegia	1
Rosaceae	Fragaria	1
Rubiaceae	Galium	1
Scrophulariaceae	Penstemon	2

This table was compiled from the list of plants found in the book “Green Roof Plants” by Edmund Snodgrass and Lucie Snodgrass (2006). Please see said book for a complete list of species as well as the requirements for each species used.