ASSESSMENT OF SMALL SCALE TALLGRASS PRAIRIE RESTORATION IN AN URBAN ENVIRONMENT

By

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THESIS ABSTRACT

Tallgrass prairie restoration is an important conservation activity in rural areas. However, little is known about prairie restoration in urban environments. The overall objective of this study was to characterize and better understand urban prairie restoration. This was carried out through an examination of 29 restoration sites within Winnipeg, Manitoba. The results indicated that actively restored urban prairies were successful and high in diversity. Multiple attributes of the restorations were examined as indicators of success including vegetation, the propagule bank and insects. However, not all attributes delivered equivocal results. This suggests that multiple measures should be used to assess a restoration site. Anthropogenic and biophysical variables were found to influence vegetation of the restorations equally, highlighting the importance of incorporating a human component in urban ecological research. These urban restorations were seen to surpass larger rural restorations in quality; thus, efforts should be made to increase their prevalence.

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CHAPTER 1: General Introduction

Urbanization is now recognized as one of the greatest threats to biodiversity (McKinney 2002*;* Liu et al 2003). The development of urban areas is steadily increasing and almost half of the earth's population and 75% of the population in developed nations now reside in cites (UNPD 2006). Urbanization exerts grave pressures on natural environments (Sharpe et al. 1986; Moffatt and McLachlan 2004), in part because cities are often located in regions of high geological and, hence, biological diversity (Kuhn 2004).

Yet, urban areas deserve recognition as important targets of conservation for protection of remnant habitat (Pautasso 2007), and, perhaps as importantly, for their role in education and increasing environmental awareness for urban residents (Miller & Hobbs 2002; Miller 2006). As cities continue to expand, it is thus likely that even degraded natural areas in the urban and suburban landscape will increase in their conservation value (Morrison et al. 1994).

Ecological restoration has emerged as an important response to the decline of natural habitat and biodiversity. It has come to be considered a critical counterpart to conservation biology (Dobson et al. 1997, Davis and Slobodkin 2004) and perhaps even "one of the most important disciplines in the whole of environmental science" (Ormerod 2003). However, most restoration is conducted in rural landscapes, and thus there is little insight into the role and potential contribution of restoration for urban environments. Although many recognize the need for ecological restoration of urban areas (e.g. Clergeau et al. 2001, Miller 2006) surprisingly little ecological research has focused on urban ecosystems of any kind (Ormerod et al. 2002). The few studies that do examine urban restorations have found that they are fraught with many challenges beyond those encountered by restorations in rural and remote areas. These include biophysical factors such as, an inability to restore natural disturbance regimes (Ehrenfeld 2000), a highly fragmented and hostile landscape matrix resulting in increased species extinction and exotic invasion (Baldwin 2004; Simenstad, Reed & Ford 2006), and isolated remnant populations (Callaway & Zedler 2004). Additionally, anthropogenic factors such as high levels of human disturbance (Grayson, Chapman & Underwood 1999; Ehrenfeld 2000) and social concerns (Gill 2005) may further complicate urban restoration.

Fortunately, the ecological and societal benefits of urban restoration are considerable as well. Restoration in urban areas can incorporate humans explicitly, resulting in unparalleled project support and long term volunteer commitment. Additionally, committed restorationists can implement site specific, small scale restoration management that often is missing from large scale rural restorations (Martin, Moloney & Wilsey 2005). The benefits of restoration also extend to the participants by reconnecting people with nature (Jordan 1994), providing increased understanding for the natural environment (Purcell, Friedrich & Resh 2002), and enhancing the quality of the urban environment (Simenstad et al. 2005).

Tallgrass prairie is recognized as one of the most endangered habitats in North America and currently accounts for less than 0.1% of its historical cover at the northern edge of its range (Samson & Knopf 1994; Samson & Knopf 1996). As such, the tallgrass prairie has become the focus of much restoration activity across North America in both rural and urban landscapes. However, the associated research has been almost exclusively rural in nature (e.g., Baer et al. 2004; Martin, Moloney & Wilsey 2005; McLachlan & Knispel 2005) and none of it has been conducted in cities. To date no systematic examination of urban prairie restoration or terrestrial urban restoration of any kind has come to light. Consequently, the overall objective of this study was to better understand the role of urban restoration for terrestrial ecosystems. This would be accomplished by examining 29 urban prairie restorations in Winnipeg, Canada. Specific objectives related to the following chapters are outlined below.

LIST OF OBJECTIVES

Objective one: To characterize the vegetation of long term urban tallgrass prairie restorations and to better understand the processes driving any associated changes (Chapter 3.). In particular, to:

- Characterize the vegetation of long term urban prairie restorations
- Assess any differences between active and passively restored sites and understand why restorations differ from each other.
- Identify the relative contribution of biophysical and anthropogenic forces on restorations in urban environments.

Objective two: To examine the success of urban tall grass prairie restoration (Chapter 4.) In particular, to:

- Contrast the success of long-term passive and active urban restoration

- Determine how success is affected by time since restoration, management intensity, and ownership;
- Assess the role of additional components of these restorations, specifically, propagule banks and insect diversity (i.e. grasshoppers and katydids), in determining restoration success.

CHAPTER 2: Literature Review

INTRODUCTION

The restoration of urban areas is an important yet understudied component of restoration ecology (Ormerod 2003). Although these habitats are highly disturbed and modified by humans, urban restoration has potential for increasing both the ecological and social value of the urban environment (Callaway & Zedler 2004). However, urban restorations may be very complex due to the altered physical environment, modified biological processes and issues resulting from the intricate social fabric of urban areas (Ehrenfeld 2000, Baldwin 2004). To facilitate further interest and increase awareness of urban restoration it is important to characterize current efforts, as well as understand the variables that influence restoration in human dominated settings.

URBAN ECOSYSTEMS

Threat of urbanization

Humans are increasingly altering their surroundings, often at the expense of biodiversity. Currently, over half of the earth's surface has been modified by humans (Vitousek 1997) and urbanization has been identified as one of the leading causes of species loss (McKinney 2006). The process of urbanization has profound impacts on many aspects of the environment influencing climate, hydrology and soils (Pickett et al. 2001). As well as the permanent disturbance associated with built environments, other impacts including feral wildlife, water runoff, waste disposal, off-roading, and trampling are all prevalent in urban areas and may result in further degradation and increased levels of exotic species (Matlack 1993, Moffatt & McLachlan. 2003). The process of urbanization eliminates most natural vegetation (Sharpe et al. 1986) and contributes to the decline of any remaining populations of native plant species (Moffatt and McLachlan 2004; Williams et al. 2005).

The replacement of a greater diversity of indigenous species with a small number of exotic species is referred to as biotic homogenization (McKinney & Lockwood 1999). Underlying this process is the tendency for opportunistic species which exhibit rapid growth, wide dispersal and are capable of breeding in ephemeral habitats to out-compete those more specialized species with slower reproduction and specific habitat requirements (McKinney & Lockwood 1999). Examples of this phenomenon are the feral pigeon (*Columba livia*) and house sparrow (*Passer domesticus*) (Dunn et al. 2006) as well as common invasive plants such as quack grass (*Agropyron repens*) and common dandelion (*Taraxacum officinale*) that have become dominant inhabitants of cities around the world.

Urban research

Despite the effects of urbanization on the ecological structure of urban areas there is a growing realization that even these ecologically “degraded” environments have an important role to play in conservation and may harbor surprisingly high numbers of species (Cornelis & Hermy 2004). Human settlements often are located in areas of high geological diversity and, consequently, higher biological diversity which further contribute to the importance of their ecological health (Kuhn 2004). Indeed, the growth of urbanized landscapes has led ecologists to identify unique processes occurring in urban environments and to recognize them as an inherently important field of study (Rebele 1994). Although urban ecosystem research has been concentrated in Europe (e.g., Kent, Stevens & Zhang 1999, Hill, Roy & Thompson 2002, Dana, Vivas & Mota 2006), examples from other regions in the world are increasing in number (e.g., Moffat & McLachlan 2004, Williams et al. 2005).

Ecological processes

Fragmentation represents a major threat to vegetation species richness and diversity in urban systems (Moffatt, Kenkel & McLachlan, 2004). Urbanization destroys much natural habitat and remaining natural areas become more isolated (Bierwagen 2007). As with rural areas, species extinction in urban fragments is related to site size, with smaller sites showing higher extinction rates (Bastin & Thomas 1999). Extinction rates also have been linked to urbanization, and fragments in urban areas have higher extinction rates than their rural counterparts (Wilson et al. 2005). Extinction events can affect vegetation directly by reducing native plant populations and seedbanks (Bastin & Thomas 1999; Moffatt & McLachlan 2004). They also act indirectly, through extinction of seed vectors such as ants (Thompson & McLachlan 2006), mammals (Mahan & O'Connell 2005) and birds (Mortberg 2001; Chace & Walsh 2006).

Ecosystems are in a state of constant flux and thus do not exist in equilibrium (Sousa 1984). Disturbance is a natural part of this dynamic process. It has been defined as a 'discrete, punctuated killing, displacement, or damaging of one or more individuals (or colonies) that directly or indirectly creates an opportunity for new individuals (or colonies) to become established' (Sousa 1984). Disturbance may include biological and physical events such as storms, fires, floods, grazing and burrowing, as well as anthropogenic events linked with construction, maintenance and recreation (Rebele 1994). In urban areas, however, anthropogenic factors are more important (Rebele 1994). Urban areas are characterized by high levels of human disturbance (van Beynen et al 2007), which affects vegetation, (Dana, Viva & Motas 2002; Moffat & McLachlan 2004; Fanelli, Tescarollo & Testi 2006) soils, (Scharenbroch, Lloyd & Johnson-Maynard 2005) birds (Mortberg 2001) and insects (Thompson & McLachlan 2007). Anthropogenic disturbances do not affect all vegetation equally. The resulting conditions tend to favour exotic species with annual life cycles (Hill, Roy & Thompson 2002; Moffat & McLachlan 2004) at the expense of perennial species (Rebele 1994).

Role in conservation

Perhaps more important than the direct role that urban areas play in conservation through habitat protection is the role they play in environmental education and awareness. The environment that people live in has profound implications for their interest in the natural world in general, and conservation in particular (Dunn et al. 2006). Half of the world's population, however, resides in biologically depauperate urban centers (UNPD 2006). The majority of urban residents are concentrated in neighborhoods of especially low biodiversity (Turner, Nakamura & Dinetti 2004). This limited interaction between people and the natural world has been found to have broad implications for conservation. Personal connections with nature are important in fostering interest in conservation (Williams & Carey 2006).

Further emphasizing the importance of biologically rich urban areas is a process which has been termed the "extinction of experience" (Pyle 1978; Pyle 2003) or "environmental generational amnesia" (Kahn 1995). This occurs as successive generations of children are exposed to progressively more degraded environments. These degraded systems unfortunately, continue to affect how people judge the ecological health of their surroundings throughout their lives. This may result in a continual lowering of the standard by which biological quality is measured, and, hence, less appreciation for the conservation of natural areas (Miller 2005).

In response, numerous authors have called for the inclusion of nature in urban areas (Miller & Hobbs 2002, Turner et al. 2004), and for enhanced opportunities for urbanites to have meaningful interactions with nature (Miller 2006). Fortunately, relatively small natural areas in urban settings may be able to function as effectively as large areas in creating appreciation for the natural environment (Pyle 2003).

RESTORATION ECOLOGY

History

The ideas behind modern restoration were first espoused by writers like Henry David Thoreau and George Perkins Marsh in the second half of the 19th

century. However, the precursors of today's restoration projects were carried out by landscape architects and horticulturalists from 1878-1940 who encouraged the use of native plants primarily for aesthetic reasons (Egan 1990). At the turn of the century, there emerged in the Midwestern United States a regionalist movement among artists and social organizers that strengthened the restoration movement. However, it was not until the 1930's that writings by Frank Waugh expressed the developing field of plant ecology as a motive for restoration. With the University of Wisconsin-Madison Arboretum project in 1934, restoration no longer belonged solely to the landscape architects and horticulturalists, but also became the domain of scientists, such as Norman Fassett and Aldo Leopold (Egan 1990).

Although science and restoration had early beginnings, it was many years before restoration ecology was recognized as a science in its own right. In 1981, William Jordan and the University of Wisconsin-Madison Arboretum published the first issue of *Restoration and Management Notes* where a key objective was to:

> "help identify restoration and management as a new discipline in its own right – an art, and perhaps a science, borrowing from other disciplines, but distinct from them in having its own aims, confronting different problems raising different kinds of question and concerned with the development of ideas and techniques particularly its own (Jordan 1981)."

The introduction of the journal *Restoration Ecology* 12 years later, made a greater effort to define restoration as a science, with a primary intent of being a venue for the sharing of scientific findings from restoration studies to a worldwide audience (Niering and Allan 1993). Early editorials in the journal suggest that "the successful restoration ecologist has, above all, to be a good scientist" (Bradshaw 1993). In response to an increasing awareness of the environmental damage humans have caused, restoration has undergone rapid growth, both as an academic discipline (Young 2000) and as a practical activity.

Description / Goals

The goal of ecological restoration may be described as "the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed" (SER 2004). An underlying assumption is that an attempt is being made to return the system to some historic state. It is thus important to realize that no system exists in stasis and, therefore, restorations should focus on restoring ecosystem processes rather than a particular suite of species (White & Walker 1997). An ecosystem may be considered fully recovered when it is able to sustain itself without further inputs (SER 2004). An important distinction must be made when using the terms *restoration* and *ecology* together. The term *ecological restoration* refers to the actual practice of restoring a project whereas as *restoration ecology* refers to the science surrounding restoration, and includes the models, concepts and theories that help facilitate the actual restoration process (Clewell 1993).

Restoration and Conservation

Despite the growing interest and application of restoration ecology, some debate remains as to its merit. One concern is that wide acceptance of restoration may waylay conservation efforts. Elliot (1982) argues that any restored system, regardless of how perfectly it replicates a non-disturbed system, is a form of forgery, and consequently, inherently less valuable. Katz (1992) further suggests that the acceptance of restorations as re-creating valued landscape may result in the commoditization of nature, whereby extant quality habitat may be degraded or destroyed under the illusion that it can easily be replaced. In response, restorationists have suggested that reservations about potential misuse of restoration should not detract from its great potential to achieve conservation goals. In fact, the optimistic approach and mindset of recovery typical of restoration ecology can provide insights into conservation (Young 2000). However, Young (2000) states that when restoration is referred to in the context of conservation biology it must be made clear that: (1) conservation (i.e. protection) of extant habitat is always favorable to restoration, and (2) potential for post-disturbance restoration provides no support for damaging existing habitat. With this understanding, restoration can be compatible with conservation biology and may become a critical part of preserving the world's ecosystems (Dobson, Bradshaw & Baker 1997). Furthermore, restoration ecology may contribute to conservation biology through providing valuable insights into the variation of nature (White & Walker 1997) and functioning as an "acid test" for ecological theory (Bradshaw 1987). Ultimately, due to the devastating effects that humans have had on the planet (Vitousek 1997) restoration may become the dominant expression of conservation by the end of the century (Young 2000).

Dichotomy between theory and practice

Restoration may be viewed in a scientific context (i.e. restoration ecology), and also in a more pragmatic, social framework (i.e. ecological restoration) (Weinstein 2007). However, there is considerable contention between these two outlooks. A number of authors hold to the premise that the naturalness and, therefore, value of a system is based on the absence of human interference. The incorporation of humans into restoration thus is self defeating (Katz 1992, Throop & Purdom 2006). Furthermore, restoration ecology emphasizes adherence to stringent scientific methods, the creation of broadly applicable theories, and peer reviewed and publishable results (Hobbs & Norton 1996; Winterhalder et al. 2004; Giardina et al. 2007). Bradshaw (1993) argues that a “successful restoration ecologist has, above all, to be a good scientist” and expresses concern that “intuition and green attitudes” will “destroy the efficiency and effectiveness of restoration ecology”. In contrast, others suggest that realistic restoration is based more on a site-specific, trial and error approach where “intelligent tinkering” is often more beneficial rather than adherence to scientific methodologies (Cabin 2007). Indeed, these researchers suggest that successful restoration may combine elements of art (Turner 1987, Higgs 1994) and suggest that the general public should be incorporated into restoration, both in defining outcomes of restoration and in project implementation (Higgs 2003, 2006).

This dichotomy runs right to the core of defining ecological restoration. The Society for Ecological Restoration (SER) has broadly described ecological restoration as “an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability" (SER 2004). However, Davis & Slobodkin (2004) suggest that restoration targets can be defined only within a social context. Restoration thus could more accurately be defined as "the process of restoring one or more valued processes or attributes of a landscape". This definition functions removes questionable attributes like *ecosystem health,* which cannot be easily defined, and allows broader more inclusive criteria to be considered when assessing restoration success (Davis & Slobodkin 2004).

Recently there has been a growing concession that the incorporation of both anthropocentric and ecocentric values in restoration is integral to its future success as an academic discipline and practical application (Purcell, Friedrich & Resh 2002; Weinstein 2007). With an increasing population and a greater urban influence on the landscape, many of the greatest challenges that restoration faces occur within urban and rapidly urbanizing areas (Higgs 2006). Thus, arguably the incorporation of people into these restorations is essential to their ecological success (Allison 2004). However, many restoration ecologists maintain that in "wilderness" areas removed from human presence, ecological health and function, rather than social value, become the primary goal (Throop & Purdom 2006; Weinstein 2007).

Passive vs. active restoration

Restoration projects may be implemented through either *passive* or *active* measures. Active restoration generally involves some form of direct human intervention to either expedite natural processes or to re-introduce missing ecosystem components. In contrast, passive restoration involves removing the external system stressors and then relying on spontaneous processes to return the system to a desired state. The latter approach has been touted as a fast (Prach 2003), self directed (Mitsch & Wilson 1996), and effective means of restoration (Ruprecht 2006) and in many cases this approach may be favoured due to the relatively low associated cost (De Steven 2006). In some situations, removal of stressors or disturbance on the environment is enough to direct the environment back to its natural state (Lee et al. 2002, Prach 2003, Ruprecht 2006, De Steven et al. 2006). Other studies, however, show only partial success (McLachlan & Bazely 2003, Middleton 2003, Kirkman et al. 2004), or failure (Laughlin 2003). In environments with long term habitat degradation, high levels of habitat fragmentation or insufficient seed sources, active intervention often is necessary for restoration to be successful (Handa & Jefferies 2000, van Diggelen & Marrs 2003).

Urban restoration

Although research in the fields of both *restoration ecology* and *urban ecosystems* have been on the rise, there has been no appreciable increase in the study of restoration in urban environments in the leading conservation and restoration journals (Figure 2.1.), despite the fact that the value of urban areas for conservation has been clearly described (Ormerod 2003).

The ecological restoration research which occurs in urban areas is dominated by aquatic systems; including wetlands (e.g., Garde et al. 2004; Zedler & Leach 1998), streams (e.g., Purcell, Friedrich & Resh 2002) and salt marshes (e.g., Gill 2005). Terrestrial systems are almost entirely absent from the literature, perhaps reflecting the assumption that these systems are too damaged to be restored with any hope for success (Grayson, Chapman & Underwood 1999). Aquatic systems may be preferentially selected for restoration due to their comparative ease of restoration (Morgan pers. com.), readily available funding opportunities, and a heightened awareness of the recreational and aesthetic potential of these systems (Purcell, Friedrich & Resh 2002; Baldwin 2004).

Urban restorations have many unique challenges which are not present in rural environments, stemming from the intimate relationships that humans have with their surroundings. Of primary importance is the high level of anthropogenic disturbance which occurs in urban areas. This may involve recreational use, trampling of vegetation, garbage disposal (Grayson, Chapman & Underwood 1999) and pollution (Ehrenfeld 2000). Furthermore, a host of social concerns may affect restoration efforts. For example, a survey of people living near a stream restoration project indicated that some residents were concerned that this activity might increase the prevalence of mosquitoes and other disease vectors; additionally, others were concerned about a decrease in safety resulting from vision-obstructing trees (Gill 2005). Another consideration is the increased cost of land acquisition in urban areas which greatly increases the cost of restoration in human dominated environments (Bernhardt & Palmer 2007). The connections of people to the pre-restored land also must be considered before urban restoration projects are undertaken. In urban areas it is important to recognize that human values may be more important than ecological function, and thus, social concerns and scientific goals must be balanced (Ehrenfled 2000; Purcell, Friedrich & Resh 2002).

In addition to the discussed anthropogenic considerations, there are biophysical barriers to the restoration of urban habitats. Many ecosystems are dependent on natural disturbances such as flooding (Ehrenfeld 2000) and fire (Howe 1994); however, the fragmented nature of urban habitats means that underlying and often necessary natural disturbance regimes may not easily be restored. The small size and patchy nature of available habitat also increases the rate of local extinction events and the likelihood of invasion by exotics (Ehrenfeld 2000; Baldwin 2004; Simenstad 2006). This is especially problematic as source populations of desirable species may be far removed from urban areas being restored (Callaway & Zedler 2004). These challenges highlight the importance of realistic expectations and an understanding of the ecological constraints for urban restoration (Grayson, Chapman & Underwood 1999; Baldwin 2004).

Despite these challenges, examples exist of successful restoration in urban areas (e.g., Baldwin 2004). Numerous ecological and social benefits abound from both the process and outcomes of these restorations. In highly degraded systems even a small restoration in an urban environment can have substantial impacts and may contribute ecological function that far exceeds its proportional size (Ehrenfeld 2000; Simenstad et al. 2005). Furthermore, there are many opportunities for mutualistic benefits that can result from social-biological interaction during urban restoration. This can be a very labour intensive activity, requiring substantial, ongoing and intensive human commitment for success (Geist & Galatowitsch 1999). Many of the most successful restoration projects are made possible by community groups and may be conceived, implemented and maintained by these groups (Gill

2005; Higgs 2005). Interaction with nature has been found to provide physiological (Ulrich 1981) and psychological (Miles, Sullivan & Kuo 2000) benefits enriching the quality of life. Restoration provides increased understanding for the natural environment (Purcell, Friedrich & Resh 2002) and enhances the quality of the urban environment (Simenstad et al. 2005).

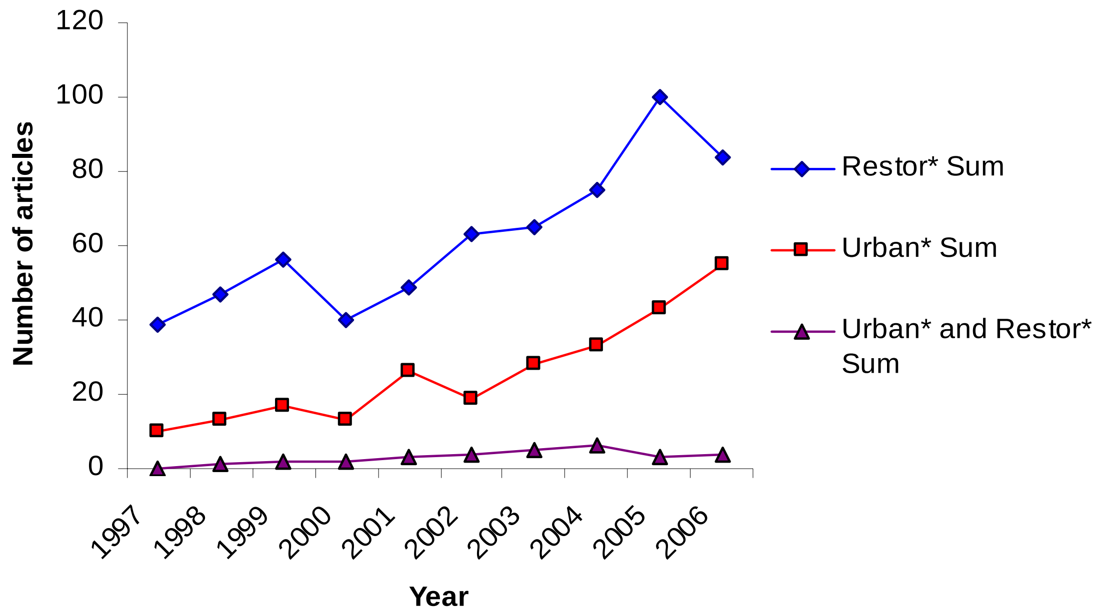

Figure 2.1. Search of applied ecology literature (i.e. Ecological Applications, Journal of Applied Ecology) and conservation literature (i.e. Conservation Biology and Biological Conservation) for submissions including the terms "restor*" and "urban*" in their abstract, title or keywords from 1997 – 2006.

Monitoring

Defining restoration goals during the planning stages of a restoration is an important step which allows efficient monitoring (SER 2004). Generally, restorations are carried out in attempt to restore at least one missing attribute of an ecosystem (Davis & Slobodkin 2004). If this is clearly defined, monitoring can be constructed to ensure restoration goals are being met. Traditionally, restoration evaluation has concentrated on using vegetation to determine restoration success and comparisons generally are made to a nearby area of extant habitat known as reference sites (Young 2000). However, a single reference site cannot adequately describe the potential trajectories a restoration can take. The use of multiple reference sites for assessment is thus preferred (Pickett & Parker 1994; White & Walker 1997).

Diversity measures such as species richness, Simpson's diversity index and effective species richness are among the most commonly used methods to assess restoration success (Ruiz-Jain & Aide 2005). They are relatively easy to measure and have been shown to influence a variety of ecosystem properties including stability (Tilman, Reich & Knops 2006), resistance to exotic species invasion (Fridley et al. 2007) and resistance to insect infestation (Wilson & Polley 2002). However, commonly used diversity measures may not incorporate abundance, and thus may oversimplify important changes in species dynamics (Brudvig et al. 2007). Consequently, numerous studies have demonstrated that management decisions should not be based solely on diversity measures (Gibson, Seastedt & Briggs 1993; McLachlan & Bazely 2003; Brudvig et al. 2007).

Ideally, ecosystem processes should be measured directly to assess restoration success. These however, are harder to measure (Herrick 2000), slower to recover and generally increase project costs (Ruiz-Jaen & Aide 2005). Ecosystem processes also may be measured indirectly. Nutrient availability, for example, may provide information on nutrient cycling (Fuhlendorf et al. 2002) and the propagule, or seed bank may provide information on seed dispersal and plant propagation (Van der Valk & Pederson 1989, Sveinson & McLachlan 2003).

Restoration studies have been criticized for their strong botanical focus (Morrison 1998). Restoration success monitoring generally is limited to vegetation, although the purpose may be to increase wildlife populations (Young 2000). Recently, many have begun to suggest that restoration studies should examine a broader range of attributes when examining restoration outcomes (Longcore 2003; Nichols & Nichols 2003; Ruiz-Jaen & Aide 2005). Arthropods are one group that has emerged as an important indicator of restoration success (Ruiz-Jaen & Aide 2005). A wide range of insects, including grasshoppers (Bomar 2001), have been used to compare restored and remnant prairies (e.g., Peters 1997; Brand & Dunn 1998). Indeed, grasshoppers and katydids (i.e. Orthoptera) may be especially useful as indicators of restoration success in prairies. They are important grazers of the prairie and often are the dominant herbivores in urban prairies. Additionally, many grasshoppers and katydids require particular food sources endemic to the prairie (Craig et al. 1999).

TALL GRASS PRAIRIE

Prairie overview

The tallgrass prairie is part of a greater grassland system which extends across central North America. Historically, this system was the largest biome in North America and extended from southern Canada to central Mexico, covering an estimated 370 million ha (Sims & Risser 2000). North American grasslands developed as the result of a rain shadow east of the Rocky Mountains, and due to periods of increased aridity which favoured grasses and forbs over trees and shrubs (Axelrod 1985). The subdivision of this broad swath of grassland into subsections has long been debated. Vegetation (Küchler 1964), soil (Wilken 1986) and insects (Hamilton 2005) have all been used as methods of classifying prairies. North American grasslands have been divided into: desert grasslands in the far south and south-west portion of the United States, California grasslands and Palouse prairie in the intermountain region west of the Rocky Mountains, pockets of mountain grasslands within the western coniferous forest, the short, mixed and tallgrass prairie of the central region, and the rough fescue prairie at the Northern limit of the grassland ecosystem (Sims & Risser 2000). Tallgrass prairie is on the eastern edge of the grassland and historically covered approximately 19% (57 000 000 ha) of the total grassland area in North America (Risser et al. 1981).

Vegetation

Tallgrass prairie is unique among the other North American grasslands in that it receives the highest amount of precipitation, and as a consequence is the most productive of all the grasslands (Bragg 1995). This region is dominated by grasses, which generally account for 80-90% of prairie biomass, yet only for 25-35% of all species richness, the rest resulting from forbs (Simms 1988). More than 200 species of vascular plants can exist in a high quality tall grass prairie, and this number can increase considerably if other habitats such as riparian areas and wetlands are included (Freeman 1998). Dominant C4 grasses include *Androgon gerardii* (big bluestem), *Andropogon scoparius* (little bluestem), *Sorghsatrum nutans* (indian grass), and *Panicum virgatum* (switch grass). The Asteraceae and Fabaceae are the two most important forb families contributing to diversity. Generally, perennial plants make up about 70% of the indigenous species richness on the tall grass prairie (Freeman 1998).

Ecological processes

The tall grass prairie is a disturbance-dependent ecosystem and fire has played an integral part in its formation and maintenance (Stewart 1951). Historically, the prairie was burned at irregular yet frequent intervals as a result of lightning and human-caused fires (Axelrod 1985; Umbanhowar 1996). Higgins (1984) however, found that the majority (73%) of the lightning fires occurred during July and August. These fires would burn for many kilometers before being stopped by rainfall or natural barriers (Leach & Givnish 1996) The effects of fire on prairie vegetation are complex and change with intensity and season of burning (Copeland, Sluis & Howe 2002). Fire influences the productivity of a prairie, selects against woody plants, and alters species composition (Howe 1994a, Howe 1994b, Howe 2000). Generally, fire tends to select for graminoids over forbs as their apical meristems are produced beneath the soil surface and are thus protected from fire (Collins & Wallace 1990).

After European settlement, fire incidence was greatly reduced, which has been attributed to active fire suppression by early settlers (Umbanhowar 1996). This led to the natural succession from prairie to savannah and forest in many areas (Robertson, Anderson & Schwatz 1997).

Historically, grazing, by bison (*Bison bison*), played an important role in the maintenance of the tallgrass prairie. Ungulate grazers affect prairie vegetation by selectively removing the dominant grasses, thus releasing subdominants from competition and causing increases in C3 grasses as well as perennial and annual forbs (Towne, Hartnett & Cochrane 2005). Grazing also prevents the accumulation of litter which allows a diversity of species to flourish (Knapp & Seastedt 1986). In the past, a complex interaction occurred between grazers and fire on the tallgrass prairie. Bison have been found to feed on areas of prairie that have been recently burned, as they preferentially select grass dominated forage over areas with a higher proportion of forbs. Thus, this targeted grazing, in combination with stochastic burning, created high levels of spatial heterogeneity on the tallgrass prairie which permitted a diversity of species to thrive (Vinton et al. 1993).

Current status

The highly productive soil and ease of mechanized cultivation has resulted in tallgrass prairie becoming the most endangered ecosystem in North America. Once covering a significant portion of North America, tallgrass prairie is now limited to highly fragmented remnants. Estimates suggest that less than 1% of its original cover remains overall, and less than 0.01% occurring at the Northern extent of its range (Samson & Knopf 1994).

Today in Manitoba, tallgrass prairie is confined to a few limited areas. A survey of the region between 1967 and 1970 by the International Biological Program found 60 areas of remaining tallgrass prairie all under 6 ha in size (Johnson 1987). The largest remnant in Manitoba is the Manitoba Tall Grass Prairie Preserve near Tolstoi and Gardenton, where 2500 ha of prairie habitat is preserved (CWHP 1998). Other smaller areas of remaining tallgrass prairie include the Saint Charles Rifle Range, the Oak Hammock and Lake Francis Wildlife Management Areas and Little Mountain Park. Two areas of remnant prairie have been protected within Winnipeg, these being the Living Prairie Museum (12 ha) and Rotary Prairie (8 ha) (WPWNS 2005). Other small fragments of prairie less than 1 ha can be found along railways and in cemeteries (Joyce & Morgan 1998).

TALL GRASS PRAIRIE RESTORATION

Establishment and Management

The process of prairie restoration has been well documented in the literature. Areas to be restored tend to be on ex-arable land (Kindscher & Tieszen 1998) or land otherwise disturbed by anthropogenic means (Montalvo, McMillan & Allen 2002). These sites are characterized by persistent exotic perennials and annual weeds and, therefore generally require some form of pre-restoration weed control. This may be achieved through removal of topsoil or solarization (Bainbridge 1990) for small-scale restorations or more commonly through a combination of herbicide spraying, plowing and disking (Wilson & Gerry 1995). For a small-scale restoration, it may be economically feasible to use transplants; however, a variety of seeding techniques including hand broadcasting, seed drilling, hydroseeding or imprinting is most commonly used (Morgan 1995).

The control of exotic weeds in the early stages of restoration is crucial (McLachlan and Knispel 2005). A wide range of methods, including overseeding (Simmons 2005), nitrogen manipulation (Morgan 1994; Wilson and Jerry 1995), and seeding into cover crops (Perry and Galatowitsch 2003), have been used with varying degrees of success.

Fire is an integral part of the tall grass prairie, reducing woody encroachment and stimulating native grasses through increased soil surface light, temperature and nitrogen availability (Hulbert 1998). Currently, fire is used extensively as a management tool for restorations and remnants. Fire season influences species dominance and persistence (Howe 1995). Dormant (i.e. spring and fall) season fires have been shown to favor C4 dominants (Howe 1995), while growing season (i.e. summer) fires may increase the presence of subdominants (Copeland et al. 2002).

Mowing is another commonly used management tool. It may be used at a small scale to simulate grazing and allow forbs to establish (Van Dyke et al. 2004; Williams, Jackson & Smith 2007) or, at a large scale, as a substitute for fire (MacDougall & Turkington 2007). Mowing is especially important as a management tool in areas such as cities, where controlled burning may be prohibited (Hitchmough & de la Fleur 2006).

While a considerable amount is known about successfully establishing prairie much remains unclear. Martin, Moloney & Wilsey (2005) suggest that current methods are still unable to restore plant diversity and that new restoration methods should be developed that utilize small-scale management activities such as mowing.

Urban prairie restoration

Numerous studies have been carried out on near-urban prairie restorations such as those at Fermilab (e.g., Sluis 2002; Lane & BassiriRad 2005) near Chicago, Illinois, and studies on the Curtis prairie at University of Madison-Wisconsin Arboretum (Sperry 1994; Allison 2002; Kucharik et al. 2006). However, these restorations are large in size. 405 ha and 28 ha respectively, near remnant habitat (Betz, Lootens & Bekker 1997; Kucharik et al. 2006) and located in areas of low population. Furthermore, none of these studies have shown any evidence of constraints imposed by surrounding urban areas. In fact, no examples of restoration of prairie or grasslands in human dominated landscapes were found.

Interestingly, the only example I found of prairie recreation in a highly urbanized environment is from Britain where prairie was created, in a region it had never historically occurred, to enhance greenspace diversity, increase aesthetic appeal and to reduce reliance on mowing for green space management (Hitchmough, de la Fleur & Findlay 2004).

STUDY AREA

Winnipeg is located at the northern extent of the tallgrass prairie ecosystem. However, the excellent soils of the region have resulted in 94 % of the surrounding land use being devoted to agriculture (Wilken 1996). Interest in prairie preservation in the province of Manitoba may be linked to a survey of the province for tallgrass prairie which was carried out in 1987 by the Manitoba Naturalists Society Tall-Grass Prairie Inventory and Conservation Project. This first systematic survey found that few areas of tallgrass prairie remained and from the representative area they surveyed they extrapolated that Mantioba's tallgrass prairie was limited to 1/20th of 1% of its pre-European settlement range, and the few areas of prairie that remained generally were limited to road right of ways, and along railroad tracks (Joyce and Morgan 1989). These alarming findings stimulated action and increased the profile of tallgrass prairie conservation in the province. This resulted in the birth of the Manitoba tall grass prairie preserve through the purchase of a 32 ha remnant of tallgrass prairie in southern Manitoba (Joyce and Morgan 1989). Since this time, a collaborative effort by both private and governmental agencies has increased the size of this original protected area to 2500 ha (CWHP 1998).

Fortunately, Manitoba's commitment to conservation has gone beyond protection of extant habitat and extended to restoration as well. The area surrounding Winnipeg is home to both Canada's first prairie restoration company, Prairie Habitats Inc. (established 1981), as well as the oldest prairie restoration in Canada which was begun at Beaudry Park in 1984. In fact, this 'green thinking' has extended to urban areas as well, through programs such as Wild Winnipeg, which has encouraged people to treat urban areas as wildlife habitat and foster biodiversity in these areas (McLachlan pers. com.). Currently, there are multiple examples of active urban prairie restorations in the city of Winnipeg which have been featured in national and international publications (e.g. Johnson 1998, Bradbury & Maddocks 2000, Primeau 2003; Lefebrvre & Fisher 2004, MNS 2007). Although many of the resulting urban restorations are small in scale and are garden like, they play a significant role in creating diverse urban habitat (McLachlan 1996). In addition to active restorations, Winnipeg hosts numerous examples of passive restoration. These often are located near existing natural areas and generally are areas that have been taken out of mowing and been allowed to undergo natural succession. Reasons for passive restorations include, enhancing the native vegetation, purifying runoff water, allowing native species to gain a stronger foothold and increasing soil quality (WPWNS 2005). Aside from an ecological benefit, passive restorations may also add structure and aesthetic appeal to areas with little natural vegetation (McLachlan pers. com.). Winnipeg thus, offers a unique opportunity to study prairie restoration in general and urban restoration in particular.

This research was conducted on 22 active tallgrass prairie restorations (i.e. where plant species were actively introduced), seven passive restorations (i.e. where no plant species were actively introduced but where surrounding management practices were modified to facilitate recovery), and seven prairie remnants within the city of Winnipeg (Fig. 2.1). The active restorations covered a broad range of efforts and included small scale in-yard restorations, those on private land, some of which were accessible to the public, and those larger public restorations carried out in parks.

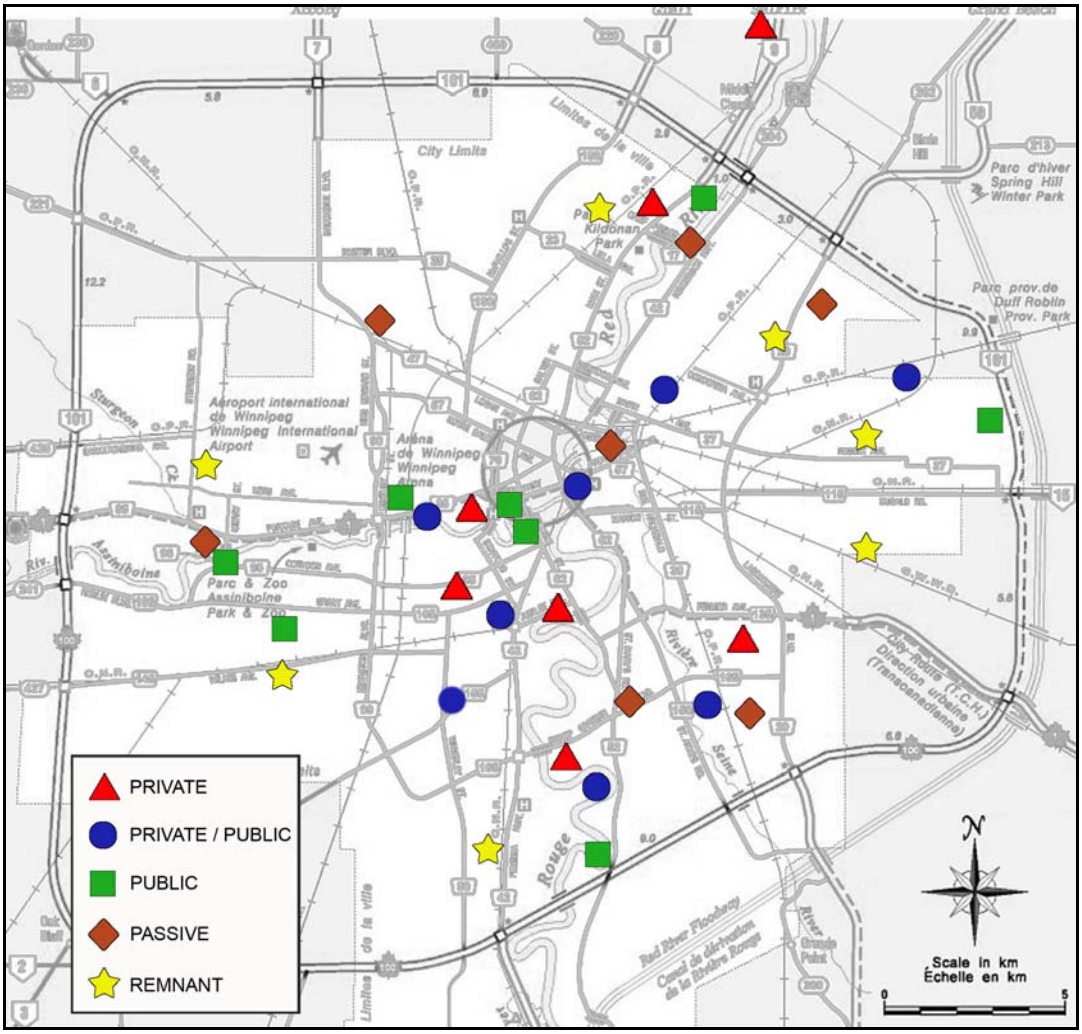

Figure 2.2. Location of active restorations (private, private/public, public), passive restorations and prairie remnants used in this study within the city of Winnipeg, MB, Canada.

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CHAPTER 3: Impacts of anthropogenic and biophysical variables on restoration

3.1 ABSTRACT

Urbanization is recognized as one of the greatest threats to biodiversity. Restoration has emerged as an important way of mitigating these declines; however, remarkably little is known about restoration in urban environments. In this study we characterized the vegetation of actively and passively restored sites relative to nearby extant prairie in Winnipeg, Manitoba. We further examined the relative roles that biophysical and anthropogenic variables play in the outcomes of these restorations. Passive restorations were characterized by low diversity of native species and a high diversity of exotics. In contrast, active restorations were highly successful and dominated by native species, although they generally had lower native diversity than remnant prairies. We found that biophysical and anthropogenic variables were equally important in their effects on the restorations; however, much of the variance described by the biophysical variables was related to anthropogenic factors. Restoration size and canopy cover were the most important biophysical variables and were negatively associated with native forb diversity. Seed mix and use of transplants were the two most important anthropogenic variables and had a substantial effect on the native vegetation of the sites. Ownership also was important and private and private/public sites were much more diverse and forb rich than public sites, despite being smaller in size and in most cases entirely dominated by edge habitat. Our results emphasize the importance of considering anthropogenic

variables in human-dominated landscapes and show that active restoration is highly successful in urban environments and worthy of further attention.

3.2 INTRODUCTION

Urbanization is now recognized as one of the greatest threats to biodiversity (McKinney 2002*;* Liu *et al* 2003). Urban development is steadily increasing and nearly half of the earth's population and 75% of the population in developed nations now reside in cites (UNPD, 2003). It is widely recognized as one of the gravest pressures on natural environments (Sharpe *et al*. 1986; McKinney 2006; Moffatt & McLachlan 2004), in part because cities are often located in regions of high geological and hence biological diversity (Kuhn 2004)

The two major threats to native plant diversity in cities are fragmentation of natural habitat and disturbance (Moffatt, McLachlan, & McLachlan 2004). These can adversely affect vegetation directly by reducing plant cover, native plant populations and seedbanks of desirable species (Bastin & Thomas 1999; Moffatt & McLachlan 2004). They also act indirectly through extinction of seed vectors such as ants (Thompson & McLachlan 2006), mammals (Mahan & O'Connell 2005) and birds (Crooks, Suarez & Bolger 2004). Aside from the intense and sustained disturbance levels in built environments, associated increases in water runoff, waste disposal, and trampling may contribute to the decline of native species and the spread of exotic species that further degrade any remaining habitat (Matlack 1993, Moffatt & McLachlan. 2003).

Yet, urban areas are important targets of conservation for protection of remnant populations (Pautasso 2007). As importantly, they help educate about conservation and increase environmental awareness (Miller & Hobbs 2002; Miller 2006). As urban centers continue to expand, it is likely that even disturbed natural areas in the urban and suburban landscape will increase in their conservation value (Morrison *et al*. 1994).

Environmental restoration has emerged as an important general response to the decline of natural habitat and biodiversity. It has come to be considered a critical counterpart to conservation biology (Dobson *et al*. 1997, Davis & Slobodkin 2004) and perhaps even "one of the most important disciplines in the whole of environmental science" (Ormerod 2003). Important in its own right, restoration can be viewed as an "acid test" for ecological theory (Bradshaw 1987). Unfortunately, there is little insight into the role and potential contribution of restoration for urban environments. Although many recognize the need for restoration of natural areas in urban environments (e.g. Clergeau *et al*. 2001, Miller 2006) surprisingly little ecological research has focused on urban restoration (Ormerod 2003).

Existing studies on urban restoration tend to concentrate on aquatic systems, including wetlands (e.g. Garde, Nicol & Conran 2004), streams (e.g. Purcell, Friedrich & Resh 2002), and salt marshes (e.g. Gill 2005). In contrast, to our knowledge, no systematic study has yet been conducted on urban terrestrial restoration perhaps reflecting the assumption that urban terrestrial ecosystems are irreparable (Grayson, Chapman & Underwood 1999). Although much restoration activity is conducted in cities (e.g. Evergreen, Society for Ecological Restoration) these activities, often referred to as "ecological restoration" tend to be community-based and both experience and outcome-oriented rather than academic in nature. Largely located in human dimensions, these urban restoration projects are often viewed as outside the purview of most ecological research (Higgs 2005).

Thus restoration research in human-dominated landscapes continues to concentrate on biophysical variables such as plant traits (Pywell *et al.* 2003), time (McLachlan & Knispel 2005), soil conditions (Pywell *et al.* 2007), grazing intensity (Martin & Wisely 2006), and connectivity (Holl & Crone 2004) without explicitly incorporating any human insights or experiences. Alternatively, urban environments can be viewed as 'socio-ecological systems', whereby ecosystems are simultaneously shaped by, and reflect ecological and social processes (Anderson 2006). As such, humans are intimately connected with and inseparable from urban landscapes and anthropogenic influences and the associated experiences and knowledge need to be explicitly incorporated into ecological thinking (Brook & McLachlan submitted).

The overall goal of this study was to explore the role of restoration for better understanding and managing urban terrestrial ecosystems. Using tall grass prairie as a case study, our specific objectives were to; (1) characterize the vegetation of long term urban prairie restorations, 2) assess any differences between active and passively restored sites and understand why restorations differ from each other (3) identify the relative contribution of biophysical and anthropogenic influences on restorations in urban environments. Tallgrass prairie is recognized as one of the most endangered habitats in North America, currently accounting for less than 0.1% of its historical cover at the northern edge of its range (Samson & Knopf 1994; Samson & Knopf 1996). What little remains continues to be degraded by human disturbance, exotics and overgrazing. Although it has become the focus of much restoration activity across North America in both rural and urban landscapes, the associated research has almost exclusively been rural in nature (e.g., Baer *et al.* 2004; Martin, Moloney & Wilsey 2005; McLachlan & Knispel 2005) and none of it has been conducted in cities.

3.3 METHODS

Study sites

We conducted our study at the northern limit of the tall grass prairie region in Manitoba, Canada, in the city of Winnipeg. The city has a population of 652,600 and covers an area of approximately 530 km^2 (Arcand *et al*. 2007) of what was once tallgrass prairie, wetland and riparian forest. Widely recognized as the location of numerous urban prairie restorations, this city is uniquely suited for this study. The soils in the area range from well to poorly drained and are in the Red River association of the Blackearth soil zone. They are underlain with lacustrine and alluvial deposits that make up the Red River Plain of the Lake Agassiz Basin (Ehlrich *et al*. 1953). The mean daily average temperature for the region is 2.6 °C and fluctuates between a mean daily maximum of 25.8 °C for July to a mean daily minimum of -22.8 °C for January. Yearly average precipitation is 513.7 mm, with 415.6 mm of this falling as rain (Environment Canada 2004).

In total, 36 sites were included in this study. Twenty-two of these were active tall grass prairie restorations (i.e. where species were deliberately introduced), seven were passive restorations (i.e. where no species were deliberately introduced but where surrounding management practices were modified to facilitate recovery), and seven were prairie remnants that acted as reference sites. Restored sites were selected to represent a diverse assemblage of prairie restorations including those that were privately owned, publicly owned and privately owned but publicly accessible. A minimum of seven sites from each of these three ownership categories were selected. In general, sites ranged in size from 29 m^2 to 26972 m^2 with a mean of

3349 m^2. All 36 sites were located within urban and suburban areas within the city limits, with the exception of a single restoration located 15 km north of the city in a suburban "bedroom" community. Indeed, we estimate that over 90% of the urban restorations in Winnipeg were included in this study.

Active restoration sites were selected for this study according to the following criteria: that they be identified by the landowner or manager as prairie restorations; that they be at least 25 m^2 in size and have an open overhead canopy; and that at least of 50% of the cover present be tallgrass prairie species native to Manitoba (according to Morgan, Collicutt & Thompson 1995). Site history and management practices differed substantially among sites. Methods of plant establishment among sites varied from seed drilling to planting with plugs. Unlike most prairie restorations located in rural areas, prescribed burns were rarely used, and only 4 (18%) of all active restorations had been burned in the last five years.

Remnant prairies in the area were characterized by a wide diversity of native plant species including the graminoids *Andropogon gerardii* (big bluestem), *Spartina pectinata* (prairie cord grass) and *Juncus balticus* (wire rush) along with native forbs *Rosa spp*.(rose) *Symphoricarpos occidentalis* (western snowberry), *Solidago spp.* (goldenrod), and *Glycyrrhiza lepidota* (wild licorice). Common exotics include *Poa pratensis* (Kentucky bluegrass), *Cirsium arvense* (Canada thistle), and *Sonchus arvensis* (perennial sow thistle).

Field survey

Vegetation was sampled over July and August 2005 for each site using a modified Daubenmire technique (Daubenmire 1959). Twenty 1 m x 1 m quadrats were randomly located and permanently marked at each of the 36 sites along transects. Due to the varied shape of the sites, the number of transects used for locating the quadrats was based on the ratio of average site length to average site width. Where the length, assumed to be the longest axis of the site, to width ratio was > 10, 20 transects were used, each containing one quadrat. If the length to width ratio was ≤ 10 and > 3, 10 transects were used, each containing two quadrats, and if the ratio ≤ 3 it was sampled with 5 transects, each containing 4 quadrats. Transects were evenly spaced and ran perpendicular to the long axis of the site. Spacing was determined by dividing the length of the site by the number of transects. The distance from the edge of the site to the position of the first transect was randomly selected between zero and the highest value that allowed proper positioning of the remaining transects. Percent cover was recorded for each species located within the quadrats in 5% increments. Uncommon species were assigned values of either 1% or 0.25% depending on abundance. A second round of sampling was conducted in May-June of 2006 to record any spring ephemerals that had been inadvertently missed in the previous year's sampling. Any species not previously noted were added to the percent cover data for the quadrats. To standardize sampling, a single observer collected all percent cover data. Nomenclature follows Scoggan (1957).

In addition, overhead canopy cover was visually estimated for each quadrat during vegetation sampling. Soil samples were collected in May and June of 2006 from 10 of the 20 permanent quadrats, these randomly identified at each site. Two cylindrical soil cores (6 cm x 6 cm) were taken from diagonal opposite corners of the quadrats and combined. Soil samples were homogenized, dried at 60 °C and sieved through a 2 mm screen to remove plant material and rock. Extraction of N, P, and K was based on the modified Kelowna test (Ashworth & Mrazek 1995). Nitrate levels were determined by NO_3-N in 2.0M KCl extracts by autoanalyzer. Phosphate analysis was carried out with the stannous chloride method. Dissolved potassium was determined by the automated flame photometry method. Extraction of sulphate followed McKeague (1978) and was analyzed using the turbidimetric method. Percent organic matter was determined by weight loss on ignition at 500 °C. A 2:1 deionized water:soil slurry was used to determine pH and electrical conductivity (McKeague 1978). Soil texture was estimated using a combination of two field tests; the moist cast test and the ribbon test (Denholm & Schut 1993).

Personal interviews were conducted with 23 landowners and managers to determine anthropogenic factors influencing restoration. A constructed questionnaire was used in all interviews and included both Likert scaled and open-ended questions. Topics of relevance to this study included restoration establishment, plant propagation, weed control, burn frequency, inputs and time commitments. Managers also provided a list of the species planted at the sites. Our study methodology was approved by the Joint-Faculty Human Subject Research Ethics Board Protocol at the University of Manitoba (#J2006:088).

Spatial variables were collected through a combination of aerial photos and ground truthing. Degree of urbanization was assessed by evaluating the area in a 200 m radius from each site using a relative scale of one through five representing intensity of human activity. At one extreme, sites surrounded by commercial and multifamily dwellings received a value of five and, at the other extreme, sites entirely surrounded by green space received a value of one. The distance to natural area was determined by measuring the length of a line from the centre of the restoration to the nearest unmown natural area, which was thought to represent potential sources of seed.

Data analysis

Species diversity measures were calculated at the quadrat level from percent cover data. Only plant species occurring in two or more sites were included in the analyses (n = 165). We calculated native, exotic and overall species diversity measures using Hill's (1973) measures. This is where N_0 (species richness) represents the total number of species, N_2 (effective species richness or ESR) is the reciprocal of Simpson's index and is less sensitive to rare species and E_3 (evenness) is calculated by dividing N_2 by the species richness. All data were (log +1) transformed to meet assumptions of normality (Sokal & Rohlf 1981), and untransformed data are presented. Differences in native, exotic and overall diversity among passive restoration, active restoration, and remnants were evaluated using one way analysis of variance (ANOVA). Where overall ANOVA models were significant ($P < 0.05$), post hoc multiple means Tamhane tests ($P < 0.05$), which allows unequal variance, were used to separate means (SPSS 2001).

Agglomerative cluster analysis of sites was carried out using Ward's method (1963) and relative Euclidean distances in PC-ORD 5.0 (McCune & Mefford 1999). Species cover values were averaged within each site. Optimal agglomeration is carried out by calculating the sum of squared deviation from the mean cluster. Sites are joined based on minimizing the increase in the error sum of squares between groups. Associations of species with identified groups of restorations were analyzed with one way ANOVA of species data. A relatively conservative significance level ($P = 0.005$) was selected due to the number of species tested ($n = 165$). Where overall model was found significant, post hoc Tukey's test ($\alpha = 0.05$) was used to separate means. Differences in native, exotic and total ESR for the defined clusters were evaluated using one way ANOVA. Where overall ANOVA models were significant ($P < 0.05$), post hoc multiple means Tamhane tests ($P < 0.05$) were used to separate means (SPSS 2001).

The relationship between environmental variables and vegetation was examined with canonical correspondence analysis (CCA) on species cover data averaged by site for all active restorations ($n = 22$). A preliminary analysis of the data was carried out with detrended correspondence analysis (DCA) using Hill's scaling and down weighting of rare species (PC-ORD, McCune & Mefford 1999). The length of the gradient, expressed in standard deviation units of species turnover, for the first axis was 4.3 suggesting that a linear approach would not be appropriate. Thus, a unimodal approach (i.e. CCA) was deemed appropriate. The default options in the program CANOCO were retained for all analyses, although rare species were down-weighted (ter Braak 1986). Variance decomposition methods (e.g., Borcard 1992; Heikkinen *et al.* 2004, Klimek *et al*. 2007) were used to discover the relative importance of anthropogenic and biophysical variables. All environmental variables were tested for significance using the forward-selection procedure of CANOCO with unrestricted Monte Carlo permutation tests (250 permutations). Two significant variables, electrical conductivity (EC) and sulphate levels were highly correlated ($r^2 = 0.81$) and thus only EC, the more significant variable, was retained in the ordination. Following recommendation by Okland & Odd (1994) only significant ($P < 0.05$) variables were included in partitioning of variance.

Variance partitioning allows a quantitative measure of the amount of variance that can be described by two or more sets of variables. It is conducted by carrying out a series of CCA and partial CCA on sets of environmental variables. Decompostion of variance followed methods described by Cushman & McGarigal (2002) and Brown *et al.* (2006) where the following steps occur: (i) a CCA of the species data by both anthropogenic and biophysical environmental variables were conducted separately; and (ii) the variance described by each set of variables independently was determined by using a partial constrained ordination where the other set of variables was designated as covariables; (iii) the difference between the partial CCA of the sets of environmental variables and the CCA where covariables were not designated represents the variance that is confounded between the two variables.

This approach allows the amount of variation explained by the environmental variables to be broken down into four components. These are: (i) the variation solely explained by anthropogenic variables; (ii) the variation solely explained by biophysical variables; (iii) the shared variation between anthropogenic and biophysical variables; and (iv) the variation in the species data which is not explained by either of these data sets.

3.4 RESULTS

Species composition of sites

The restored sites were generally high in diversity. In total, 230 vascular plant species were identified in 29 restored and seven reference sites. Of these, 35 (15%) were native graminoids, 140 (61%) were native forbs, and 55 (24%) were exotic species.

The exotic, *Poa pratensis* (Kentucky bluegrass) was the most abundant and widespread species, occurring in 92% of sites and averaging 16 percent cover at each site. The highly desirable native C4 grass *Andropogon gerardii* (big bluestem) was the second most abundant, making up 9.7% of the groundcover of sites. The five most frequently occurring species with their frequencies in declining order of importance were the exotics *Poa pratensis* (Kentucky bluegrass, 92%), *Cirsium arvense* (Canada thistle, 92%). *Taraxacum officinale* (common dandelion, 86%), *Agropyron repens* (quack grass, 83%) and *Vicia cracca* (tufted vetch, 72%). Vegetation was well established at all sites, and, on average, only 10% of each site was bare ground and 18% was covered with leaf litter. In general, restorations were more fertile than the remnants and had higher levels of nitrate and phosphate. There were few differences between restored and reference sites for other soil variables (Table 3.1).

Diversity

Species diversity differed significantly among reference sites, active restorations and passive restorations. Overall species richness and native species richness was significantly ($P < 0.001$) highest for the remnant sites, followed by the active restorations (Table 3.2). However, effective species richness (ESR) did not differ significantly between the remnants and active restorations. The passive restorations had significantly lower levels of native species ($P < 0.001$) and significantly higher exotic species richness ($P = 0.008$) and ESR ($P = 0.013$) than either active restorations or reference sites. Native species evenness was significantly highest in active restorations and lowest in the passive restorations ($P < 0.001$). No differences were detected in overall evenness and exotic species evenness among sites ($P = 0.056$, $P = 0.245$, respectively). A comparison to a reference data set from nearby rural prairie remnants (from McLachlan & Knispel 2005) indicated that the urban remnants were very similar in quality to rural remnants; however, they had a slightly higher native species ESR and lower ESR of exotic species.

Species composition

There were clear differences in the vegetation composition of different sites. Five clusters were selected based on agglomerative cluster analysis, accounting for 40% of the variability in the data. Passive restorations along with one low quality restoration (Plate 3.1) separated out from the active restorations and the reference sites (Fig 3.1, Plate 3.2) The actively restored sites then further divided into forb-dominated sites (Plate 3.3), grass-dominated sites (Plate 3.4), and sites with a mix of grass and forbs (Plate 3.5). These five clusters were clearly evident in the biplot depicting Axis 1 and Axis 3 of the Correspondence Analysis (CA) (Fig 3.1). Axis 2 (data not shown) represented an effective habitat-quality gradient separating out reference sites from active and passive restorations.

Table 3.1. Description of urban tall grass prairie restorations (passive and active) and reference sites. Significance is indicated by *P < 0.05.

	Passive restoration	Active restoration	Reference
Number of sites	7	22	7
Growing seasons	12.57 †	7.96 ± 0.83	N/A
Area (hectares)	0.57 ± 0.36	0.24 ± 0.08	0.45 ± 0.20
Shade (%) (trees)	1.18 ± .69	8.24 ± 2.84	0.82 ± 0.74
Leaf litter (%)	20.19 ± 1.47	16.17 ± 3.06	23.46 ± 2.26
Bare ground (%)	1.21 ± 0.65	14.88 ± 2.89*	2.00 ± 1.21
Nitrate (ppm)	3.71 ± 1.22	7.68 ± 2.59	2.16 ± 0.48
Potassium (ppm)	569.14 ± 19.07	562.41 ± 15.60	528.33 ± 56.09
Phosphorous (ppm)	34.37 ± 10.21	47.04 ± 3.54*	5.57 ± 1.26*
Sulphate (ppm)	13.14 ± 2.44	10.36 ± 1.25	11.83 ± 1.92
H+ ion concentration	1.64 ± 0.21	1.85 ± 0.13	2.2 ± 0.41
Organic matter (%)	5.97 ± 0.69	6.54 ± 0.46	5.95 ± 0.23
EC (dS/m)‡	0.95 ± 0.12	0.75 ± 0.03	0.80 ± 0.19

† Estimated average age of passive restorations

‡ Electrical conductivity

Table 3.2. Mean Hill's measures (±SE) for tall grass prairie restorations and reference sites compared to measures from nearby rural remnants.

	Passive restoration	Active Restoration	Urban reference	Rural Remnant *	P - value
Species richness					
Native	0.46 ± 0.15 [c] †	4.94 ± 0.54 [b]	7.76 ± 0.57 [a]	6.95 ± 0.42	0.000
Exotic	4.33 ± 0.37 [a]	3.01 ± 0.24 [b]	2.23 ± 0.23 [b]	2.02 ± 0.17	0.008
Overall	4.76 ± 0.35 [c]	7.96 ± 0.52 [b]	9.99 ± 0.64 [a]	8.98 ± 0.49	0.000
Effective species richness					
Native	0.36 ± 0.12 [b]	2.83 ± 0.29 [a]	3.44 ± 0.22 [a]	3.18 ± 0.21	0.000
Exotic	2.33 ± 0.14 [a]	1.83 ± 0.12 [b]	1.37 ± 0.07 [c]	1.41 ± 0.08	0.007
Overall	2.43 ± 0.13 [b]	3.87 ± 0.27 [a]	4.21 ± 0.26 [a]	3.91 ± 0.26	0.005
Evenness					
Native	0.27 ± 0.07 [c]	0.63 ± 0.02 [a]	0.47 ± 0.02 [b]	0.50 ± 0.03	0.000
Exotic	0.57 ± 0.02	0.63 ± 0.02	0.69 ± 0.04	0.81 ± 0.04	0.194
Overall	0.56 ± 0.02 [a]	0.50 ± 0.02 [a]	0.44 ± 0.04 [a]	0.46 ± 0.03	0.034

* Data shown for comparison from McLachlan & Knispel (2005), collected with similar methods within 20 km of urban area used in this study. Data not incorporated into calculation of P - values.

† Means followed by the same letter lack statistical significance (P <0.05).

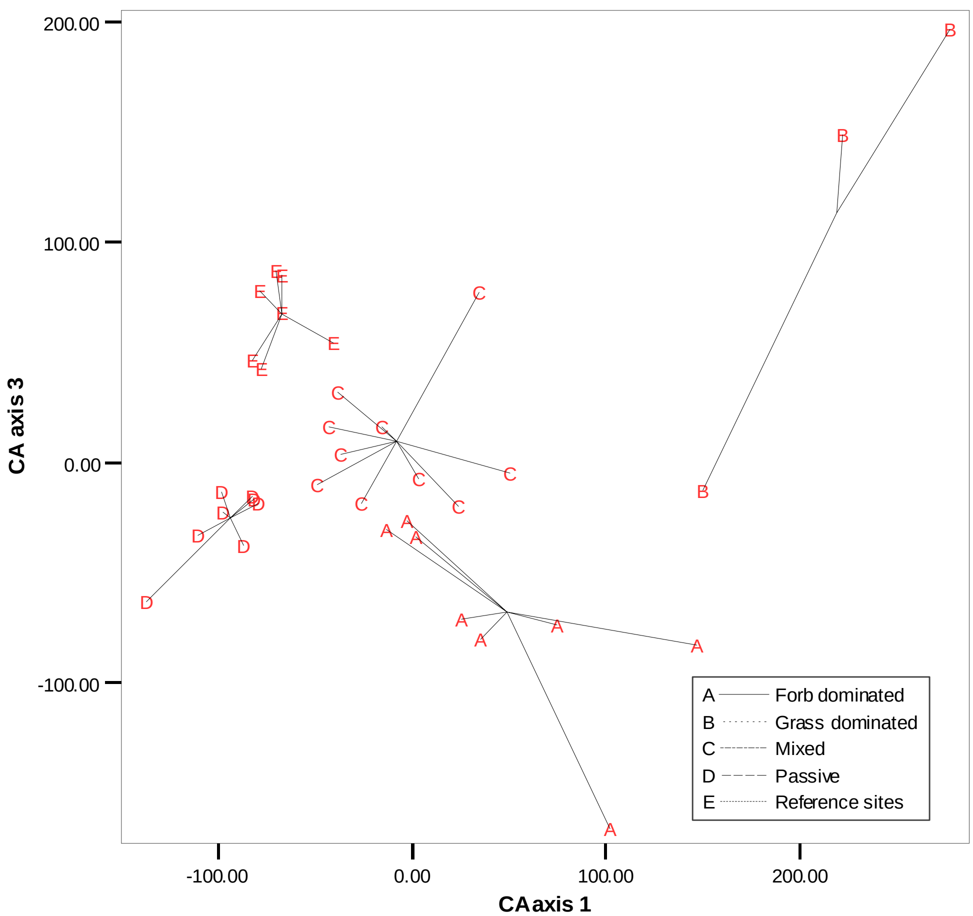

Figure 3.1. Canonical analysis (CA) of all restoration sites showing groupings according to cluster analysis.

Plate 3.1. Example of passive restoration effort in Winnipeg, MB (i.e. Cluster D).

Plate 3.2. Example of reference site in Winnipeg, MB (i.e. Cluster E).

Plate 3.3. Example of forb dominated restoration in Winnipeg, MB (i.e. Cluster A).

Plate 3.4. Example of grass dominated restoration in Winnipeg, MB (i.e. Cluster B).

Plate 3.5. Example of mixed restoration in St. Andrews, MB (i.e. Cluster C).

Table 3.3. Summary of plant species showing a significant response to groupings determined with cluster analysis. Life form and origin shown and data are presented as mean percent cover (±SE). Means followed by different letters are significantly different.

Species	Life form *	Origin †	Forb dominated (A)	Grass dominated (B)	Mixed (C)	Passive (D)	Remnants (E)	*P*-value‡
Agropyron smithii	C3	N	-	2.76 ± 1.53 [a]	0.03 ± 0.03 [b]	-	-	0.000
Agropyron trachycaulum	C3	N	-	2.22 ± 0.42 [a]	-	-	-	0.000
Agrpryron subsecundum	C3	N	0.12 ± 0.12 [b]	3.27 ± 1.11 [a]	0.08 ± 0.06 [b]	0.15 ± 0.10 [b]	-	0.000
Bromus inermis	C3	E	0.58 ± 0.31 [b]	-	1.92 ±1.13 [b]	16.42 ± 5.49 [a]	0.29 ±0.21 [b]	0.000
Deschmapsia caespitosa	C3	N	0.07 ± 0.07 [b]	0.81 ± 0.68 [a]	-	-	-	0.002
Juncus balticus	C3	N	-	-	-	-	2.48 ± 0.98 [a]	0.000
Poa pratensis	C3	E	10.42 ± 4.70 [b]	0.72 ± 0.72 [b]	14.57 ± 3.27 [b]	29.42 ± 4.00 [a]	20.46 ± 2.41 [ab]	0.000
Andropogon gerardii	C4	N	1.35 ±0.53 [b]	8.63 ± 1.82 [a]	24.65 ± 5.27 [a]	-	14.58 ± 4.08 [a]	0.000
Bouteloua curtipendula	C4	N	0.25 ± 0.16 [b]	4.32 ± 1.32 [a]	0.28 ± 0.22 [b]	-	-	0.000
Panicum virgatum	C4	N	0.52 ± 0.23 [b]	6.90 ± 3.51 [a]	0.57 ± 0.31 [b]	-	-	0.000
Schizachyrium scoparium	C4	N	2.08 ± 0.10 [ab]	2.58 ± 1.47 [a]	1.22 ± 0.83 [b]	-	0.04 ± 0.04 [b]	0.002
Sorghastrum nutans	C4	N	0.29 ± 0.19 [b]	2.58 ± 1.47 [a]	0.08 ± 0.06 [b]	-	-	0.000
Spartina pectinata	C4	N	0.03 ± 0.03 [b]	-	0.03 ± 0.03 [b]	-	6.11 ± 2.40 [a]	0.000

Table 3.3. Continued

Species	Life form *	Origin †	Forb dominated (A)	Grass dominated (B)	Mixed (C)	Passive (D)	Remnants (E)	*P*-value‡
Agastache foeniculum	Forb	N	1.47 ± 0.61 [a]	0.20 ± 0.20 [ab]	0.16 ± 0.11 [b]	-	-	0.002
Aster novae-angliae	Forb	N	1.98 ± 0.77 [a]	0.20 ± 0.20 [ab]	0.58 ± 0.40 [b]	-	-	0.001
Cirsium arvense	Forb	E	2.02 ± 0.94 [b]	-	4.02 ± 0.86 [ab]	8.74 ± 2.20 [a]	2.20 ± 0.57 [b]	0.002
Heliopsis helianthoides	Forb	N	3.63 ± 1.29 [a]	0.72 ±0.72 [ab]	0.36 ± 0.30 [b]	-	-	0.001
Monarda fistulosa	Forb	N	2.21 ± 0.83 [a]	0.50 ± 0.50 [ab]	0.29 ± 0.15 [b]	-	-	0.001
Polygonum erectum	Forb	N	-	0.09 ± 0.09 [a]	-	-	-	0.010
Thalictrum venulosum	Forb	N	-	-	-	-	0.20 ± 0.10 [a]	0.004
Veronicastrum virginicum	Forb	N	1.18 ± 0.44 [a]	0.09 ± 0.09 [ab]	0.18 ± 0.11 [b]	-	-	0.002
Populus tremuloides	Woody	N	-	-	0.06 ± 0.06 [b]	0.03 ± 0.03 [b]	0.52 ± 0.21 [a]	0.001
Rosa spp.	Woody	N	0.34 ± 0.27 [b]	-	0.12 ± 0.10 [b]	0.03 ± 0.03 [b]	4.32 ± 1.98 [a]	0.000
Symphoricarpos occidentalis	Woody	N	-	-	-	0.12 ± 0.12 [b]	2.39 ± 0.93 [a]	0.000

* C3 = graminoid with C3 photosythetic pathway, C4 = graminoid with C4 photosynthetic pathway

† N = Native, E = Exotic

‡ *P*-value of overall model statement

Results of an ANOVA on species data indicated that 24 of the 165 species differed significantly ($P < 0.005$) among these five clusters (Table 3.3). The first class (i.e. actively restored sites that are dominated by forbs) had high abundance of the native forbs *Agastache foeniculum* (giant hyssop), *Aster novae-angliae* (new-england aster), *Monarda fistulosa* (bergamot) and *Veronicastrum virginicum* (culver's root) as well as the highest levels of exotic forbs (Table 3.3, Table 3.4). In contrast, the second class (i.e. actively restored sites that are dominated by grasses) had low exotic diversity and was dominated by native C3 and C4 grasses including *Panicum virgatum* (switch grass), *Sorghastrum nutans* (indian grass), and *Agropyron spp.* (wheat grasses). The third class (i.e. actively restored sites with mix of grass and forbs) was characterized by *Andropogon gerardii* (big bluestem) and *Cirsium arvense* (Canada thistle) and had the lowest species diversity out of the actively restored sites (Table 3.3). The fourth class (i.e. passively restored sites) had low levels of native species and high levels of exotic *Bromus inermis* (smooth brome), *P. pratensis* (Kentucky bluegrass), and *Cirsium arvense* (Canada thistle). These passive restorations, as well as the prairie remnants, had the highest levels of *P. pratensis* (Kentucky bluegrass). The fifth and final class (i.e. remnant prairies) had significantly more woody species than the other sites, including *Populus tremuloides* (trembling aspen), *Symphoricarpos occidentalis* (western snowberry), and *Rosa spp.* (wild rose). These remnants also had the highest native forb diversity.

Table 3.4. Effective species richness (±SE) of graminoids and forbs separated by origin for restoration groupings. *P* value of overall model statement shown.*

	Forb dominated (A)	Grass dominated (B)	Mixed (C)	Passive (D)	Remnants (E)	P -value
Native						
Graminoid	0.81 ± 0.11 [c]	4.03 ± 0.81 [a]	1.14 ± 0.14 [b]	0.11 ± 0.14 [d]	1.257 ± .094 [b]	<0.001
Forb	2.78 ± 0.37 [b]	1.20 ± 0.56 [c]	1.48 ± 0.29 [c]	0.44 ± 0.15 [d]	3.36 ± 0.23 [a]	<0.001
Total	4.15 ± 0.30 [b]	5.30 ± 0.38 [a]	3.34 ± 0.42 [c]	2.46 ± 0.11 [d]	4.21 ± 0.26 [b]	<0.001
Exotic						
Graminoid	1.02 ± 0.19 [b]	0.57 ± 0.13 [c]	1.18 ± 0.10 [b]	1.58 ± 0.98 [a]	1.06 ± 0.05 [b]	<0.001
Forb	1.57 ± 0.09 [a]	0.70 ± 0.25 [b]	1.41 ± 0.20 [a]	1.50 ± 0.16 [a]	0.90 ± 0.14 [b]	<0.001
Total	1.94 ± 0.07 [b]	1.07 ± 0.13 [d]	1.97 ± 0.22 [b]	2.26 ± 0.13 [a]	1.37 ± 0.07 [c]	<0.001

* Means followed by different letters are significantly different at ($P < 0.05$).

Table 3.5. List of anthropogenic and biophysical variables collected for tall grass prairie restorations (*n* = 22) and corresponding accumulated sums of canonical eigenvalues from forward selection procedure. *P* value of each variable presented.

Variable	*P* value	Accumulated sum of canonical eigenvalues
Anthropogenic		
Seed mix ratio of grass species to forb species	0.004*	0.38
Percentage of transplants used	0.004*	0.62
Public ownership	0.024*	0.82
Intensity of exotic species control	0.040*	0.98
Private / public ownership	0.048*	1.13
Thatch clearing	0.016*	1.25
Floristic quality of planted species	0.092	-
Native control	0.084	-
Burn frequency	0.498	-
Length of site preparation	0.677	-
Level of urbanization	0.096	-
Private ownership	0.250	-
Biophysical		
Area	0.020*	0.26
Canopy cover	0.036*	0.51
Age	0.024*	0.74
Electrical conductivity	0.048*	0.92
Percent clay	0.012*	1.09
Percent sand	0.020*	1.22
Nitrate	0.096	-
Organic matter	0.104	-
Phophorous (ppm)	0.681	-
Distance to natural area	0.797	-
H+ concentration	0.769	-

* indicates significance at P < 0.05

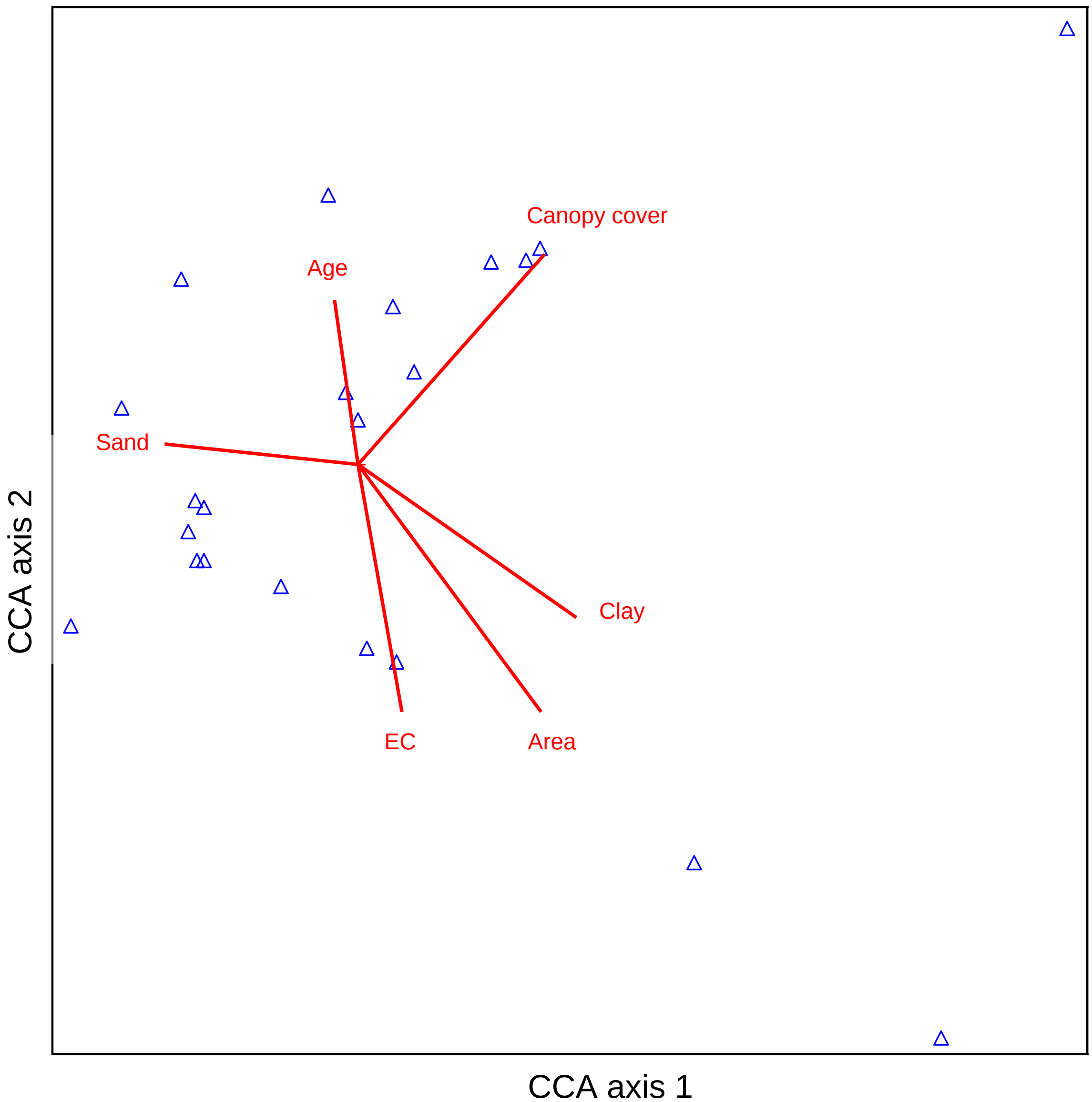

Figure 3.2. Results of partial Canonical Correspondence Analysis (CCA) of active urban restoration (n = 22) vegetation constrained by significant biophysical variables (n = 6 variables shown on 2x scale). Variable abbreviations used are: Sand = percentage of sand in soil, Clay = percentage of clay in soil, EC = electrical conductivity.

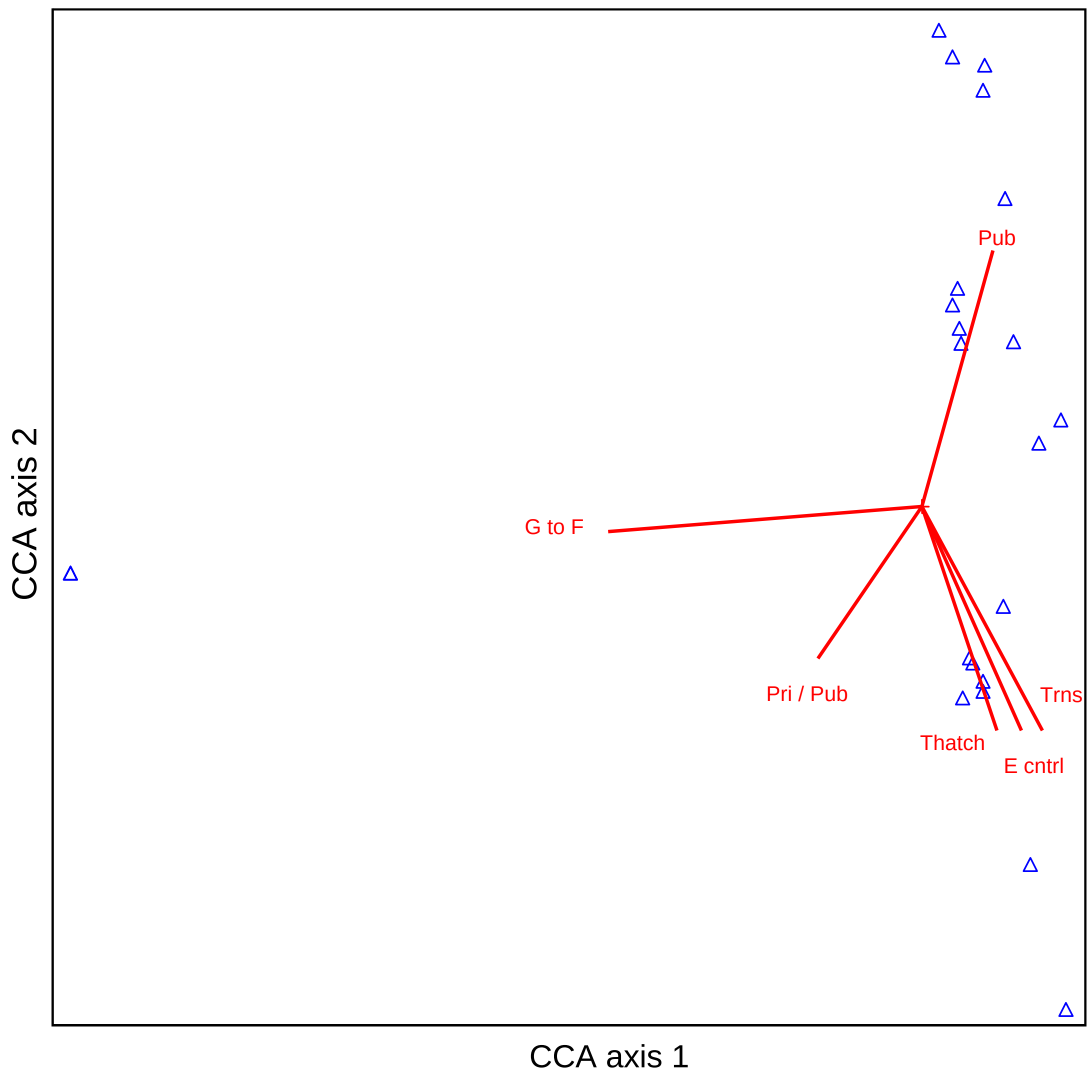

Figure 3.3. Results of partial Canonical Correspondence Analysis (CCA) of active urban restoration sites (n = 22) vegetation constrained by significant anthropogenic variables (n = 6 variables shown on 2x scale). Variable abbreviations used are: G to F = seed mix ratio of grass species to forb species, Pub = public ownership, Trns = percentage of transplants used, E cntrl = intensity of exotic species control, Thatch = clearing of thatch, Pri / Pub = Private / Public ownership.

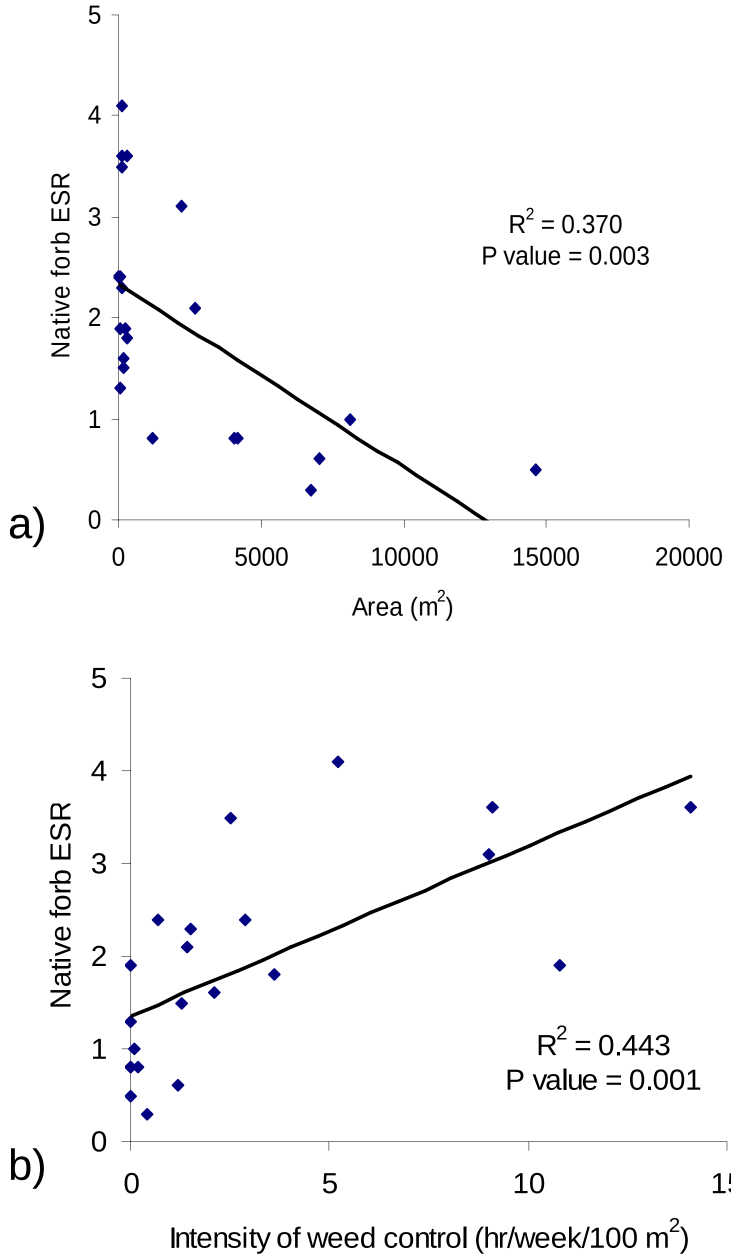

Figure 3.4. Relationship between native forb effective species richness (ESR) and: (a) area; and (b) intensity of weed control (n = 22).

Biophysical and anthropogenic variables

Biophysical and anthropogenic environmental variables explained a large amount (66.4%) of the variation in the species data. The total inertia (sum of all unconstrained eigenvalues) in the species data was relatively high at 3.22. Forward selection of significant environmental variables resulted in six biophysical and six anthropogenic variables being retained for subsequent constrained ordination (Table 3.5). Twelve percent of the variance was confounded between the biophysical and anthropogenic variables, and 33.6% was unexplained by either set of variables.

The biophysical variables explained 26.6% of the overall variance. Restoration area was the most important of these variables accounting for 21% of the total variation (Table 3.5). Native grass species such as *Agropyron smithii* (western wheat grass), and *Beckmania syzigachne* (sloughgrass) along with the exotic forb *Melilotus spp.* (sweet clover) and native forb *Dalea purpureum* (purple prairie clover) were associated with larger sites. Restoration area also had a strong negative influence on native forb diversity ($P = 0.003$, $r^2 = 0.370$) (Fig. 3.4a) and was inversely related to canopy cover; the next most important biophysical variable (Fig. 3.2). Sites with substantial canopy cover were characterized by shade tolerant species such as exotic *Glechoma hederacea* (ground ivy), *Arctium minus* (common burdock) and native *Parthenocissus quinquefolia* (virginia creeper). Restoration age accounted for 19% of the model variance and was associated with native woody species such as *Populus tremuloides* (trembling aspen), and *Prunus virginiana* (chokecherry) and perennial exotics such *Medicago sativa* (alfalfa) and *Sonchus arvensis* (perennial sow thistle). Restoration age was not significantly associated with percent cover of native graminoids (P = 0.828) but positively associated with exotic graminoid cover (P = 0.004, r^2 = 0.334 data not shown). The three soil variables: electrical conductivity, percentage of clay and percent sand explained the remaining 39% of model variance the latter two variables predictably and inversely related (Table 3.5). In general, sites with a higher percentage of sand were small intensively managed restorations whereas larger sites tended to have a higher percentage of clay, likely reflecting soil amendments that were largely carried out in small areas. Percentage of sand was found to be generally related to presence of native forbs such as *Anemone patens* (prairie crocus) and *Potentilla fruticosa* (shrubby cinquefoil).

Anthropogenic variables accounted for 27.8% of the overall variance and six of the 12 collected variables were significant. The two most important anthropogenic variables were both related to site revegetation and explained a combined 50% of the model variation (Table 3.5). These were the seed mix (i.e. ratio of seeded grass species to forb species) and the percentage of each site revegetated with transplants. The ratio of seeded grass to forbs dominated axis 1 and clearly separated out two sites restored using a seed mix completely dominated by grass species (Fig. 3.3). Public ownership was the third most important anthropogenic variable explaining 16% of the variation and was positively associated with axis 2 of the CCA (Fig. 3.3). Species associated with public sites included exotic *Trifolium repens* (white clover) and ruderal natives such as *Asclepias speciosa* (showy milkweed). Axis 2 was negatively associated with management variables including weed control, annual thatch clearing, and transplant use. These sites were associated with most strongly associated with conservative species such as *Bouteloua gracilis* (blue grama grass), *A. patens* (prairie crocus), and *Festuca hallii* (rough fescue). Intensity of weeding was the fourth most important variable and described 13% of the variation in the anthropogenic CCA. This variable, which was thought to be an indicator of management, strongly influenced native forb diversity (P = 0.001, r^2 = 0.443, Fig. 3.4b). The final two variables, corporate ownership and annual clearing of thatch described 12% and 10% of the variation, respectively (Table 3.5).

3.5 DISCUSSION

Generally, active prairie restoration was highly successful in these urban environments. Although the restorations were usually quite small, averaging 0.24 ha in size, they were characterized by high native graminoid and forb diversity. As with other, albeit non-urban, studies (e.g. Sluis 2002, Martin, Maloney & Wilsey 2005; McLachlan & Knispel, 2005), we found that the restorations generally had lower native diversity and higher exotic diversity than comparable prairie remnants. However, there was relatively little difference in diversity measures between the extant and restored prairies in our study. Prairie restorations usually have 30-80% of the diversity of extant prairie (Martin, Maloney & Wilsey 2005; McLachlan & Knispel 2005; Polley 2005), whereas the actively restored sites we examined approached 90% of the diversity of the extant prairies. This is likely a function of the relative high quality of the restorations, rather than the poor quality of the (urban) remnants as urban and rural remnants had largely comparable species diversity values (McLachlan & Knispel 2005).

The diversity of some restorations approached that of extant prairies; nevertheless, there were considerable differences in species composition between the restorations and remnants. The limitations of using diversity measures as measures of success are now well recognized and it is increasingly common that other factors such as species abundance and ecological processes are examined when restoring natural habitat (McLachlan & Bazely 2001, 2003; Ruiz-Jaen & Aide 2005; Brudvig *et al*. 2006; Chapter 4). Although many sites had a high diversity of graminoids, many forbs were present. Indeed, a subset of small and private sites (i.e. forb cluster) were characterized by high levels of desirable native forbs including *A. foeniculum* (Giant hyssop)*, A. novae-angliae* (new-england aster)*,* and *Heliopsis helianthoides* (oxeye sunflower). The prevalence of these sexually reproducing species contrasts with previous research in rural restorations where seed-dependent forb populations declined and often were extirpated over time as C4 grasses rapidly became dominant (Baer *et al*. 2004; Polley 2004; McLachlan & Knispel 2005).

Our results did show, however, that a number of the larger, generally publicly managed active restorations (i.e. mixed restoration cluster) tended to follow the dynamics typical of rural restorations. These restorations were characterized by low native forb diversity and dominated by native C4 grasses and by the exotics *C. arvense* (Canada thistle) and *B. inermis* (smooth brome). These patterns are likely associated with the under-management that often hampers inadequately resourced public restorations (Chapter 4). Although fire is rarely used as a management tool for urban restorations, all four of the restorations that had been burned in the past five years fell into this mixed restoration cluster, suggesting that spring burning in urban sites might benefit dominant and warm season graminoids at the expense of native forbs.

Contrary to other studies (e.g. Prach 2003; De Steven 2006), our findings demonstrated that passive restoration was unsuccessful at recovering the species diversity of native grasslands (Chapter 4). These sites were dominated by exotics such as *C. arvense* (Canada thistle) and ruderal native species; suggesting that passive restoration of grasslands in urban environments is unlikely to ever become dominated by native species. This reflects the near absence of a native propagule bank in these longstanding green spaces and, in turn the relatively short propagule viability of many of the prairie species (Bossuyt & Hermy 2003; Laughlin 2003). It further suggests that the distance of restorations from other prairie habitat and the hostile intervening matrix likely compromised seed dispersal in this urban environment.

Biophysical and anthropogenic variables

Much of the ecological literature on restoration has focused on the biophysical factors that might affect changes in species composition and richness (e.g., Dzwonko 1994; Bartolome *et al.* 2004). Substantially less is known about how human management practices, or anthropogenic variables, influences vegetation in restorations. Even less is known about how anthropogenic variables influence restorations in urban centers.

Time since restoration has a substantial effect on species composition and dominance of prairie restorations. Other studies find that species richness, particularly of forbs, tends to decrease over time (McLachlan & Knispel 2005) whereas cover of native perennial grasses often increases during this period (Baer *et al*. 2002; Sluis 2002; Martin & Wisely 2006). These changes may occur due to competitive advantages of C4 grasses resulting from high pre-restoration nutrient levels (Baer *et al*. 2002; 2004), an over-reliance on spring burns for management (Copeland, Sluis & Howe 2002), and the rhizamatous nature of the dominant prairie grasses (Collins & Wallace 1990). Although species composition did change over time in this study (Chapter 4), cover of native graminoids and/or forbs showed little change. In contrast, woody species such as *Populus tremuloides* (trembling aspen) and *Prunus virginiana* (chokecherry) were associated with older sites, suggesting that woody species become established in the absence of management practices that counter these changes, a common finding in grasslands (e.g., Ansley & Castellano 2006; Brudvig *et al*. 2007). The perennial weeds *Sonchus arvensis* (perennial sow thistle) and *Medicago sativa* (alfalfa) also were associated with older sites. Although these results contrast with Mclachlan & Knispel (2005) who found that exotic species generally declined over time, they attributed this decline to the burn-associated dominance of C4 grasses, which did not occur in this study.

Urban habitat is often characterized by low light levels relative to that found in rural regions, this attributed to the built environment and the presence of shade trees on many city streets (Akbari, Rose & Taha 2003). Species associated with elevated canopy cover in our study included shade tolerant exotic forbs such as *Arctium minus* (common burdock) and *Glechoma hederacea* (creeping ground ivy). These C3 forb species may gain a competitive edge against C4 prairie grasses that are photosynthetically less efficient at low light levels (Turner & Knapp 1996; Awada *et al*. 2003). Moreover, leaf litter produced by these large trees may further bolster the growth of invasive species (Siemann & Rogers 2003). Nutrient levels under tree canopies have been found to increase relative to surrounding grasslands through leaf deposition and decomposition, favouring fast growing exotic plants (Scholes & Archer 1997, Averett *et al*. 2004) and causing decreases in grassland species richness (Klimek *et al.* 2007). In fact, canopy cover was a limiting factor in our study. Many landowners wanting to restore natural habitat in older neighbourhoods were often forced to establish plant communities that were characteristic of forest understoreys rather than prairies because of excessive shade from established trees.

Patch size had a substantial effect on the outcomes of restoration in this study. It is widely recognized that plant populations are more vulnerable to extinction in small habitat remnants, in part because of the proportional increase of edge habitat and their vulnerability to disturbance (Bastin & Thomas 1999) and changes in microclimate (Saunders, Hobbs & Margules 1991). The restorations in our study were small enough that most consisted entirely of edge habitat. Although restoration area was identified as the most important biophysical variable affecting vegetation; it is likely that these differences in species composition were indirect in nature and more likely due to management strategies that were affected by the size of the restorations. Larger sites were generally planted with seed drills and not transplants, which may favour the establishment of grasses (Morgan pers. com.). Moreover, larger sites usually were publicly owned, whereas the smaller sites generally were privately owned, which in turn had implications for the intensity of vegetation management as is discussed below.

Anthropogenic variables were as important as biophysical variables in their influence on these urban restorations. These human influenced variables operated directly on the vegetation and, as just described, indirectly, through their effects on the biophysical environment. Overall, the anthropogenic influences seemed to be most important during site establishment.

Seed mixes had an important influence on vegetation during restoration. Although the similarity between sown species composition and subsequently observed species composition was typically quite low (Chapter 4), the proportion of seeded grass-to-forb species and percentage of transplants used were important predictors of vegetation. The grass-to-forb ratio separated out two sites that had been seeded almost entirely with grass species, along with *D. purpureum* (purple prairie clover), a narrow leaved forb. This allows for the large-scale application of selective broad-leaf herbicides such as 2,4-D during site establishment and the elimination of problem weeds including *C. arvense* (Canada thistle), *S. arvensis* (perennial sow thistle), *Euphorbia esula* (leafy spurge), and *Melilotus spp.*(sweet clovers). This use of herbicide tolerant species in the early stages of restoration and the subsequent addition of forbs once weed control and the dominant native grasses have become established might represent an effective way of controlling exotics early on in restoration (Brown and Bugg 2001). However, it is not yet clear how readily these subsequently introduced forbs will become established, especially in relatively isolated sites without viable propagule banks (Brown & Bugg 2001). Others have even questioned whether forbs need to be added at all as they represent such a minor component of the vegetation and because forb seed is usually less accessible and affordable than that of grasses (Bakker *et al.* 2003). However, with the majority of tallgrass prairie diversity arising from forbs and not graminoids, forbs are an integral component of prairie restorations (Turner & Knapp 1996). Additionally, forbs are generally more attractive to urban restorationists than graminoids (Kotyk 2007; Benvie & McLachlan 2005), especially those in private sites, and thus graminoid-dominated restorations are of questionable social value in urban environments. These problems are compounded by a widespread distrust of pesticides by many of the managers we interviewed (Kotyk 2007). The use of pesticides was seen as undermining the conservation potential of the restorations. Indeed, beyond the social value of forbs, these species have been found to play a valuable role in reducing weed invasion of restorations through their contribution to grassland diversity (Brown & Bugg 2001).

The degree to which transplants were used to establish sites had a substantial impact on the species composition of the restorations. Transplants represent a more expensive method of restoring vegetation that was commonly used for all species on smaller sites and occasionally for forb species on larger sites. Transplants may be advantageous for forb species allowing them a competitive advantage over established grass seedlings (Brown & Bugg 2001). Although more expensive, transplants have a greater rate of survival and allow for greater control, enabling managers to match plants with microclimates at very fine scales and add species which do not develop well from seed in the field (Morgan, Collicutt & Thompson 1995).

Site management varied among ownership classes, in turn affecting species composition (Higgins *et al.* 2002). Publicly owned sites tended to have higher levels of persistent perennial and annual exotic species and showed the highest rates of desirable species loss over time (Chapter 4). Limited resources often result in public weed control programs that focus only on conspicuous weeds such as *C. arvense* (Canada thistle), that require control under provincial noxious weeds control by-laws

(Sveinson pers. com.). In contrast, private sites and private/public sites were higher in native graminoid and forb diversity and had lower rates of desirable species loss (Chapter 4) in large part because they were so small.

The small size of most urban restorations and the active involvement of site managers have positive implications for the control of invasive species control and ultimately for the native diversity of these sites. We found, perhaps counter-intuitively, that native forb diversity was highest in small restorations and actually declined as restoration size increased. Although larger urban sites tended to follow the vegetation dynamics of rural sites, with high levels of C4 native grasses and lower levels of forb populations (Baer *et al.* 2002, McLachlan & Knispel 2005) the small urban restorations remained forb rich because of the active management and commitment demonstrated by gardeners. Many rural studies suggest that finer scale site management and weed control is important (e.g. Holl & Crone 2004; Martin, Maloney & Wilsey 2005; McLachlan & Knispel 2005). In our study, larger sites with low levels of weed control were strongly associated with invasive species such as *Sonchus arvensis* (perennial sow thistle), and *Bromus inermis* (Smooth brome) whereas private and private/public sites were characterized by highly motivated and better resourced managers and more intensive weed control. Aside from immediate increases in ecological integrity resulting from the removal of exotics, control of invasive species may release desirable and otherwise uncommon plants from competition. This may facilitate an increase in native species abundance and result in higher native diversity, especially of forbs.

Conclusions and management implications

This study has demonstrated how inextricably linked humans and natural habitats become in human-dominated landscapes. Although clearly relevant for rural and even remote environments, this is especially true for densely populated urban landscapes. These urban areas are dominated by built environments and by green spaces that normally are managed to reduce or eliminate diversity. Moreover, any extant forest and grassland in urban centers continues to be threatened by development and subsequent invasion by exotics (Moffat, McLachlan & Kenkel 2004; Williams *et al*. 2005). The role of restoration in these environments is only beginning to be explored. Importantly, our results indicate that, when left to generate naturally, passively restored sites showed little recovery over time, and remained low in diversity and dominated by exotics. Thus, active management of restorations, indeed remnants, is essential for conservation of prairie in urban environments. While these outcomes are important in their own right, they might help explain why urban ecology and, more specifically, urban restoration is largely absent from the academic literature.

In North America, this shortfall might reflect a continued preoccupation with high integrity and even "wilderness" areas in ecological research (see Katz 1993) and thus rural and especially urban systems are largely seen as devoid of ecological value (Higgs 2005). Although there is a growing recognition in Europe of the importance of diverse and increasingly threatened agro-pastoral grasslands (e.g., Pywell *et al*. 2002; Dauber, Bengtsson & Lenoir 2006) and, to a lesser degree, urban habitat (Goode 1989; Hitchmough & de la Fleur 2006), this is still largely absent from the North American literature. These gaps might point to the shortcomings of ecological theory, much of which reflects systems that have little if any human presence (Miller 2002). It is perhaps not surprising that this work affords less insight for systems where humans play an important, and, in urban systems, the key ecological role. Human experiences and values are rarely referred to, much less actively incorporated in ecological research (Brook & McLachlan submitted). To our knowledge there has yet to be an urban restoration study that does so.

Yet, our results indicate that restorationists played a substantial and deliberate role in shaping these prairie habitats that was equivalent in importance to the biophysical factors that otherwise dominate the literature. Moreover, many of the biophysical factors (e.g. area, canopy cover) are themselves linked to these human-mediated factors. That human preferences may play a defining role in the composition of urban restorations helps to explain the surprising prevalence and diversity of forbs in many of the restorations in our study in comparison to other restoration studies in rural environments. Managers showed a sustained and intimate involvement with these restorations and often selected plant species that were aesthetically pleasing or that resonated with their cultural values (Kotyk 2007).

In conclusion, the outcomes of this study indicate that these urban restorationists were highly successful at generating species-rich prairie habitats. Although most of the restorations were quite small, many contained high value prairie plants, including threatened species such as *Veronicastrum virginicum* (culver's root) and *Aster sericeus* (western silvery aster) that are listed under the provincial Endangered Species Act (CSSM 2006). Similar to wildlife refuges, these relatively small urban prairie restorations may function as depositories of genetic diversity for highly threatened species, thus allowing the persistence and possible re-introduction of these species into areas where they have long been extirpated. In fact, the small size and active involvement of people in these restorations may be the key to their success allowing levels of knowledge and management unachievable in larger rural restorations. Their value in turn explicitly reflected the motivations of the restorationists, especially for the privately owned sites. The managers were at once cognizant of the precarious status of tall-grass prairies and the conservation value that these urban restorations represented (Kotyk 2007). As well as supporting important plant species, urban restorations provide for many invertebrate species including *Danaus plexippus* (monarch butterflies), *Papilio polyxenes* (black swallow tails) and *Speyeria spp* (fritillaries), these also of concern to many managers (Mutch pers. obsv.). However, the educational value of these sites also was recognized by the managers, allowing residents to experience an otherwise threatened ecosystem which most of them had yet to encounter. The combined ecological, conservation, and education values of these sites strongly supports the role of restoration in urban centers. The remaining challenge is to find ways of increasing these initiatives, which can only enhance their visibility and access to human and non-human residents alike.

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CHAPTER 4: Urban tallgrass prairie restoration: determinants of success

4.1 ABSTRACT

Tallgrass prairie is one of the most threatened ecosystems in North America and less than 0.1% of its historical cover remains at its northern limits. Restoration is recognized as an important response to this decline and the processes and outcome of this restoration has been widely studied in rural areas. In contrast, the numerous restoration projects occurring in cities have yet to be the focus of any systematic study. We investigated the success of urban restoration for 29 sites in Winnipeg, Canada. Included were active restorations, where desirable plant species were introduced, and passive restorations, where no species were introduced but surrounding management was modified to facilitate recovery. The diversity and species composition of aboveground vegetation were examined, as well as those of the grasshoppers and katydids (Orthoptera) and the propagule bank. Our results indicated that active restorations were high in diversity and became more similar to reference sites over time; however they lost desirable native species as they aged. In contrast, passive restorations were low in diversity and were dominated by exotic species. Intensive weed management, as seen on active restorations, had a positive effect on the floristic quality of sites and reduced the rate of species loss. Many of the differences among urban restorations were explained by ownership patterns. Both private and private/public restorations were substantially higher in forb diversity and floristic quality than public restorations. However, the propagule banks of the restorations were lower in abundance and diversity of native rhizomes than reference sites. In contrast, the active restorations had a higher abundance of

native seed. The diversity of Orthoptera increased with grass percent cover and sward height. Our results showed that urban prairies were successfully restored through active management and that multiple measures should be used to evaluate restoration success. Indeed, the outcomes of these active restorations are often more successful than their rural counterparts and have much conservation and education value. The challenge now is to increase the ubiquity of these restorations.

4.2 INTRODUCTION

Ecosystems across North America have been dramatically changed over the last 150 years due to intensive urbanization and agriculture (Vitousek et al. 1997). Tallgrass prairie is one of the most threatened of these as less than 1% of its total former cover, and less than 0.01% at its northern limits, remains (Samson & Knopf 1994; Samson & Knopf 1996). The restoration of degraded prairies has emerged as a logical and important complement to habitat protection (McLachlan & Knispel 2005) and is generally carried out in rural landscapes (e.g., Martin, Moloney & Wilsey 2005; Copeland, Sluis & Howe 2002).

In some cases, grasslands are passively restored, whereby removing external stressors is enough to set the habitat to on a trajectory towards its pre-disturbance state (e.g., Prach 2003, Ruprecht 2006). However, seedbanks in these rural landscapes tend to be devoid of desirable native species (Sveinson & McLachlan 2005) and sites are usually isolated from extant prairie. Thus, these habitats are generally actively restored; where the existing seedbank is depleted by repeated mechanical cultivation and herbicide application and native plants are subsequently reintroduced (Morgan, Collicutt & Thompson 1995).

An important determinant in the outcome of these active restorations is the time since the site was restored (Baer et al. 2002). Although rural restorations generally become more like extant habitat over time (Kindscher & Tieszen 1998; McLachlan & Knispel 2005), long-term restorations remain very different from nearby remnants (Allison 2002). Often restorations become dominated by native graminoids and exotics to the exclusion of native forbs. This reflects the highly disturbed surrounding landscape as well as the reliance of managers on spring burns to control early season invasive species (McLachlan & Knispel 2005).

Most restoration is conducted on publicly owned sites, yet human-dominated landscapes are predominately privately owned. For this reason, private land has substantial unrealized value for conservation. Ownership also has important implications for conservation (Higgins, Larson & Higgins 2001); however, the implications of private ownership on ecological processes remains poorly understood (Norten 2000). This is especially true for degraded habitat located in or around cities.

Urban areas continue to grow in size and concentration, now representing more than 80% of the population in North America (UNPD 2006). However, the great majority of research on prairies, and natural habitat as a whole, is still conducted in rural and remote regions (Miller & Hobbs 2002). While there is much restoration activity in cities, this is often conducted by community stewardship groups and is practical and outcome-oriented in nature and thus it remains absent from the ecological literature (Higgs 2003). This, in part may reflect an assumption on the part of many biologists that these concrete-locked habitats are unsalvageable (Grayson, Chapman & Underwood 1999) and also may reflect the inadequacy of much ecological theory for urban landscapes. Yet, urban areas are increasingly recognized as important targets of conservation efforts (Kuhn 2004; Pautasso 2007), and, arguably represent “unparalleled” educational and social value (Box & Harrison 1994). Restoration studies that have been conducted in urban areas are principally aquatic (e.g., Zedler & Leach 1997; Grayson, Chapman & Underwood 1999; Garde et al. 2004) or riparian (e.g., Fullerton et al. 2006) in focus, and none, to our knowledge, has been conducted on an upland terrestrial system, much less the tallgrass prairie.

A primary goal of restoration is the re-creation of a self-sustaining ecosystem (SER 2004); however, assessments of restorations are often limited to the vegetation, perhaps under the assumption that if the vegetation is established remaining ecological processes will follow (Young 2000; Longcore 2003). The role and relative success of non-target components of restoration is a much neglected component of restoration. Examinations of ancillary ecosystem components, such as insects (Bisevac & Majer 1999; Longcore 2003; Blakely et al. 2006) and birds (Ruiz-Jaen Aide 2005b), are beginning to show that they recover more slowly than those components that are directly restored. It is thus increasingly recognized that more than one group of organisms should be examined, and ideally at least one that is not the target of the restoration efforts (Nichols & Nichols 2003, Ruiz-Jaen Aide 2005).

We investigated outcomes of urban tallgrass prairie restoration in Winnipeg, Canada. Specifically, our objectives were to: (1) Characterize the success of long-term passive and active urban restoration; (2) determine how success is affected by time since restoration, management intensity, and ownership; and (3) assess the role of ancillary components of these restorations, specifically propagule banks and the Orthoptera (grasshoppers and katydids), in determining restoration success.

4.3 METHODS

Study area

We conducted our study at the northern limit of the tallgrass prairie in Manitoba Canada. As a province, Manitoba contains virtually the entire tallgrass prairie region located in Canada (Samson & Knopf 1994). Historically, this ecosystem covered 6000 km^2 (1.1%) of Manitoba; however, agriculture, woody encroachment and urbanization have reduced its cover to less than 0.01% of its former range (Joyce & Morgan 1989). With a population of 652 600, Winnipeg is located in the heart of the Canadian tallgrass prairie region, and covers an area of approximately 530 km^2 (Arcand et al. 2007). This city is widely recognized for its urban prairie restorations (Johnson 1998; Bradbury & Maddocks 2000; Primeau 2003; Lefebrvre & Fisher 2004; MNS 2007). The soils in the area range from well to poorly drained and are in the Red River association of the Blackearth soil zone. They are underlain with lacustrine and alluvial deposits that make up the Red River Plain of the Lake Agassiz Basin (Ehrlich et al. 1953). The mean daily average temperature for the region is 2.6 °C and fluctuates between a mean daily maximum of 25.8 °C for July to a mean daily minimum of -22.8 °C for January. Yearly average precipitation is 513.7 mm, with 415.6 mm of this falling as rain (Environment Canada 2006).

Study sites

In total, 36 urban prairie sites were used in this study, and were initially located with the assistance of private and public agency staff and local restoration businesses. Twenty-two of the sites were active tallgrass prairie restorations (i.e.

where plant species were actively introduced), seven were passive restorations (i.e. where no plant species were actively introduced but where surrounding management practices were modified to facilitate recovery), and seven were prairie remnants. Of those that were actively restored, a minimum of seven sites from each of the three following ownership categories were selected: private, public, and privately owned but publicly accessible restorations. Sites ranged in size from 29 m^2 to 26 972 m^2 with a mean of 3 393 m^2. All 36 sites were located within urban and suburban areas within the city limits, with the exception of a single restoration located 15 km north of the city in a suburban "bedroom" community. In fact, over 90% of the urban restorations in Winnipeg were included in this study.

We selected active restoration sites according to the following criteria: they were identified by the landowner or manager as prairie restorations; they were at least 25 m^2 in size and had an open overhead canopy; and at least of 50% of the cover present be tallgrass prairie species native to Manitoba (according to Morgan, Collicutt & Thompson 1995). Site history and management practices differed substantially among sites. Methods of plant establishment among sites varied from seed drilling to planting with plugs. An average of 32 (± 4.07) species was planted at each site; the highest number of planted species was 61, and the lowest, eight species. Unlike most prairie restorations, prescribed burning was rarely used, and only four (18%) of all actively restored prairies had been burned in the last five years.

Remnant prairies in the area were characterized by a wide diversity of native plant species including graminoids such as big bluestem (*Andropogon gerardii*), prairie cord grass (*Spartina pectinata*) and wire rush (*Juncus balticus*) along with the forbs wild rose (*Rosa spp*.), western snowberry (*Symphoricarpos occidentalis*), goldenrod (*Solidago spp.*), and wild licorice (*Glycyrrhiza lepidota*). Common exotics include Kentucky blue grass (*Poa pratensis*), Canada thistle (*Cirsium arvense*), and perennial sow thistle (*Sonchus arvensis*).

Field survey

We sampled vegetation over July and August 2005 for each site using a modified Daubenmire technique (Daubenmire 1959). Twenty 1 m x 1 m quadrats were randomly located and permanently marked at each of the 36 sites along transects. A second round of sampling was conducted in May - June of 2006 to record any spring ephemerals that had been inadvertently missed in the previous year's sampling. Species not previously noted were added to the percent cover data for the quadrats. Due to substantial differences in site shape and size, the number of transects used for locating the quadrats was based on the ratio of average site length to average site width. In all cases the quadrats were randomly located on each transect. Where the length, assumed to be the longest axis of the site, to width ratio was > 10, 20 transects, with one quadrat located on each, were used. If the length to width ratio of the site was ≤10 and > 3, 10 transects, with two quadrats on each, were used and if ≤3 the site was sampled with 5 transects with 4 quadrats on each. Transects were evenly spaced and ran perpendicular to the long axis of the site. The distance from the edge of the site to the first transect was randomly selected. Cover was recorded for each species rooted within the quadrats in 5% increments.

Uncommon species were assigned values of either 1% or 0.25% depending on abundance. Percent cover of leaf litter and bare ground was also recorded. To standardize sampling, a single observer collected all percent cover data. All native species observed at each site while walking established transects were recorded to create a species inventory for the sites. Nomenclature and origin (i.e. native vs. exotic) follows Scoggan (1957).

We conducted individual interviews were with 23 restoration owners and managers to determine anthropogenic factors influencing restoration. A questionnaire was used that contained both Likert scaled and open ended questions. Topics covered included restoration establishment, plant propagation, weed control, burn frequency, inputs and time commitments of restoration. Managers also provided a list of the species planted at the sites. Our methodology was approved by the Joint-Faculty Human Subject Research Ethics Board Protocol at the University of Manitoba (#J2006:088).

Propagule bank

In October and November of 2005, we collected propagule bank samples. Two soil samples from opposite corners of every second of the 20 vegetation quadrats were taken from each site. Soil was collected to a depth of 5 cm with a bulb planter. Total volume from each sample was approximately 190 cm^3. The two samples from each quadrat were mixed, resulting in 10 samples per site (n = 36). A 300 cm^3 sub sample was then spread on top of 1 cm of sterile growth medium (Pro-Mix 'BX') 5 cm x 15 cm pots and placed under 6280 K full spectrum lights to grow out propagules. The soil was exposed to a 16-hr photoperiod beginning and watered as needed. Emerging seedlings and sprouts were identified, categorized as rhizome or seed, and removed. Where sprouts emerged from underground vegetative structures such as taproots, they were categorized as rhizomes to avoid a third category. If identification was not immediately possible they were transplanted to another pot for further growth and subsequent identification. After 21 weeks, germination had ceased and the soil was put into a cold room at 3 °C. After six weeks, the samples were removed, stirred and placed back under grow lights for an additional 18 weeks. By this time germination had again ceased and the study was terminated.

Survey of grasshoppers and katydids

We documented the use of restorations and remnants by grasshoppers and katydids, collectively referred to as Orthoptera, for a subset (n = 25) of the study sites. Five sites from each of the passive restorations, private restorations, private/public restorations, public restorations and reference prairies were examined. Three rounds of sampling (July, August, and September) were carried out during the summer to allow detection of species with different phenological patterns. All sampling occurred between 9:00 and 17:00 on sunny days (< 15% cloud cover) without strong winds (≤ 25 km/hr) following sampling criteria of Kemp, Harvey and O'Neill (1990). Five previously established transects, from the vegetation study, at each site were walked for a distance of 5 m and swept 10 times each, resulting in 50 sweeps per site. Sweeps consisted of a 180° arc through the vegetation with a standard sweep net (38 cm diameter) as described by Evans (1984, 1988) and were alternated between a sweep close to the ground and sweeps near the top of the vegetation (Narisu & Schell 2000). We placed samples collected via sweeping into plastic bags and put them on ice for transportation. Upon arrival at the laboratory they were frozen and subsequently identified. We determined density of Orthoptera by walking 5 m transects while holding three 1m pieces of PVC pipe joined in a “U” shape with the unattached ends pointed forward (Samways 1990). Transect walks have been found to be accurate in areas with low Orthoptera density (< 2 adults / m^2) (Gardiner, Hill & Chesmore 2005). All grasshoppers and katydids flushed from the 5 m x 1 m section sampled were recorded. Species that flushed and landed further down the transect line were not recounted. Average sward height was estimated for sites in August using a categorical scale of one through five.

Data analysis

Species diversity measures were calculated at the quadrat level from percent cover data. Only plant species occurring in two or more sites were included in the analyses (n = 165). We calculated native, exotic and overall species diversity measures for each site using Hill’s (1973) measures. Where *species richness* represents the total number of species, *effective species richness* (ESR) is the reciprocal of Simpson’s index and is less sensitive to rare species and *evenness* is calculated by dividing ESR by the species richness. Diversity data were (log +1) transformed to meet assumptions of normality (Sokal & Rohlf 1981). Percent cover data was determined at the quadrat level and arcsine transformed for normality before analysis (Zar 1996). In all cases, untransformed data are presented. Differences among restoration strategy (i.e. passive and active) and restoration ownership were evaluated for above ground vegetation and the propagule bank using one way analysis of variance (ANOVA). Where overall ANOVA models were significant ($p < 0.05$), post hoc Student-Newman-Keuls tests (SNK) ($p < 0.05$), were used to separate means (SAS Institute 2002).

Restoration success was determined, in part, using Sorensens' similarity index and a more conservative modified Chao-Sorensen index (Chao et al. 2005). The similarity between the observed species composition and the planted species composition was compared for each site using Sorensen's similarity index. Which is determined from the formula $2a/(2a + b + c)$ where a is the number of species shared between the planted and observed species composition, b represents the number of unique species to one species list and c represents the number of species unique to the other species list (McLachlan & Knispel 2005). Because Sorensen's similarity index is based on presence/absence data we used our complete species inventory for the sites rather than species list derived from quadrat sampling. Similarity between restorations and remnants was determined using a modified Chao-Sorensen index. This modified index was chosen as it is abundance-based and thus less sensitive to sample size and rare species (Chao et al. 2005). All similarity measures were computed using the program EstimateS (Colwell 2006).

Floristic quality assessment was used as an indicator of restoration success. This method has been shown to be better at distinguishing differences between sites than traditional richness indices (Bourdaghs, Johnston & Regal 2006). It is carried out by assigning a value from 1- 10, known as a Coefficient of Conservatism (C), to each of the native taxa found in the sampling unit. The number represents the affinity for the species to differing habitat qualities where species ranked 0-1 are found in areas of severe disturbance, and species ranked 9-10 are thought to be found only in high quality natural areas (Taft 1997; Brudvig et al. 2007). Coefficient of Conservatism (C) values are area specific and are generally determined by panels of experts who together assign conservatism values to species. We used values assigned by the Northern Great Plains Floristic Quality Assessment Panel (NGPFQAP 2001). A number of indices of quality may be derived from the floristic quality assessment procedure; however, we selected the more detailed weighted Floristic Quality Index (FQI) (Bourdaghs, Johnston & Regal 2006) as this has increased descriptive power over a traditional FQI (Poling, Banking & Jablonski 2003). This is expressed as:

$$wFQI = wC(\sqrt{N})$$

Where N is the total number of all species found at the site and wC is an abundance based Coefficient of Conservatism. This is calculated as the product of the proportional abundance *(p)* and the C value of the jth species summed for all species (N) (Bourdaghs, Johnston & Regal 2006).

$$wC = \sum_{j=1}^{N} p_j C_j$$

Following from other studies, floristic quality was calculated at the site level (Bourdaghs; Johnston & Regal 2006, Taft, Hauser & Robertson 2006).

To compare the effects of management on restoration outcomes, we created a maintenance index for each active restoration based on information collected from interviews with site managers. Most management activities were related to weed control and included in the index as hours/week of weeding/1000 m^2. Other activities such as annual clearing of thatch and control of undesirable dominant native species were thought to increase the likelihood of restoration success and were included. These two activities were weighted equally and were simply assigned values of one if they occurred. The annual addition of transplants to the restorations was included in the index as a categorical value. If a high level of plant additions occurred (≥ 10/year) a value of two was assigned; if less than 10 plants were added a value of one was assigned; where no plants were added no increases were awarded.

We examined the relationship between restoration outcome measures, age and maintenance. The measures: weighted floristic quality, similarity to planted, and similarity to remnants, were all plotted against age of restoration and management index using linear regression (SPSS 15). When similarity to planted species composition was plotted against management index, we removed the young restorations (< 4 years of age) from this analysis as their similarity to the planted species composition was predictably high (Fig. 4.2a), thus confounding the relationship between management and similarity.

To better understand changes in species composition over time we created a survivability index for all of the commonly planted species (planted at four or more sites). Survivability was defined as the number of sites where a particular species was observed divided by the number of sites where it was planted. The complete species inventory from the sites was used for these calculations.

4.4 RESULTS

Site preparation of the urban prairie restorations was achieved and dependent on a combination of mechanical cultivation and herbicide use (Fig 4.1). In contrast, the revegetation of these sites and the management of invasive species reflected the small size of these restorations. Planting of live plugs was the most common method of revegetation followed by hand broadcast of seed. Weed control was carried out mainly through hand pulling, which represented 80% of the total weed control effort (Fig. 4.1).

Species Diversity

The urban restorations examined in this study were generally high in diversity. In total, we identified 229 vascular plant species at 29 restored and seven reference prairies. The majority (76%) were native in origin and included 140 forbs and 35 graminoid species. The remaining 55 (24%) species were exotics. In general, graminoids were more abundant than forbs, accounting for 66% of the overall vegetation cover in the restorations. The exotic, *Poa pratensis* (Kentucky bluegrass) was the most abundant and widespread species, occurring in 92% of sites and covering on average 16% of each site. Yet, the highly desirable native C4 grass *Andropogon gerardii* (big bluestem) was the second most abundant species, making up 9.7% of the groundcover of sites.

Although the restorations generally were species rich, the reference prairies were more diverse. Both the active and passive restorations had significantly lower ($p = < 0.0001$) native species richness and higher exotic species richness ($p = <$

0.0081) than the reference sites (Table 4.1). In general this was found to be true for effective species richness (ESR) of both exotic ($p = 0.0068$) and native ($p < 0.0001$) species. The reference prairies also had significantly higher native species cover ($p < 0.0001$) and less exotic cover ($p = < 0.0001$) than the restorations (Table 4.1).

Restoration strategy also had significant effects on the vegetation. The active restorations had significantly higher native diversity ($p = < 0.0001$) and lower levels of exotics ($p = < 0.0068$) than the passively restored sites (Table 4.1). The passive restorations were dominated by exotics, and on average 98% of the ground cover was exotic in origin, compared to 40% for the active restorations.

Influence of time and management

Time since restoration had a substantial influence on the restoration outcomes. The overall similarity of the restorations to the remnant prairies increased significantly ($p = 0.0023$) over time (Fig. 4.2b), in part, because planted species tended to disappear. Of the 91 most frequently planted species (planted at four or more restorations), 38 (42%) were found at less than 50% of the sites in which they had been planted. In fact, one particularly degraded site contained no planted native species. The similarity between species that had been planted and those that were observed decreased significantly ($p = 0.0029$) over time (Fig 4.2a). Recently restored sites (i.e. ≤ 5 years old) more closely reflected the list of planted species than those that were older (≥ 10 years old), having similarity values of 0.564 and 0.346, respectively. The species with low or very low survivability were more conservative (i.e. *Anemone patens*, prairie crocus) species ($p = 0.0024$), these having average coefficient of conservatism values of 7.13 compared to 5.51 for the species with average or good survivability (Fig. 4.3). However, there was no significant relationship between the floristic quality of sites (FQI) and age ($p = 0.3107$, data not shown).

Site management had a substantial impact on the outcomes of restoration. When, older sites (> 4 years old) were examined, the similarity of species that were observed to those that had been planted increased significantly ($p = 0.0232$) with management intensity (Fig. 4.4a). The floristic quality index of these sites increased strongly ($p < 0.0004$) with management intensity (Fig 4.4b). However, there was no relationship between management intensity and the similarity of restorations to remnant prairies ($p = 0.6453$; data not shown), this, in part, reflecting anthropogenic planting preferences.

Species planted in restorations thus often differed in species composition from that of the prairie remnants. Of the five most frequently occurring planted native grasses, two species, *Panicum virgatum* (switch grass) and *Sorghastrum nutans* (indian grass), were absent from all the urban remnants as well as from nearby high quality rural prairie remnants (Table 4.2). Similarly, the most frequently occurring native forb at active restorations, *Aster novae-angliae* (new-England aster) and other commonly occurring forbs such as *Agastache foeniculum* (giant hyssop) and the threatened *Veronicastrum virginicum* (culver's root) were not found in any of the remnants (Table 4.2).

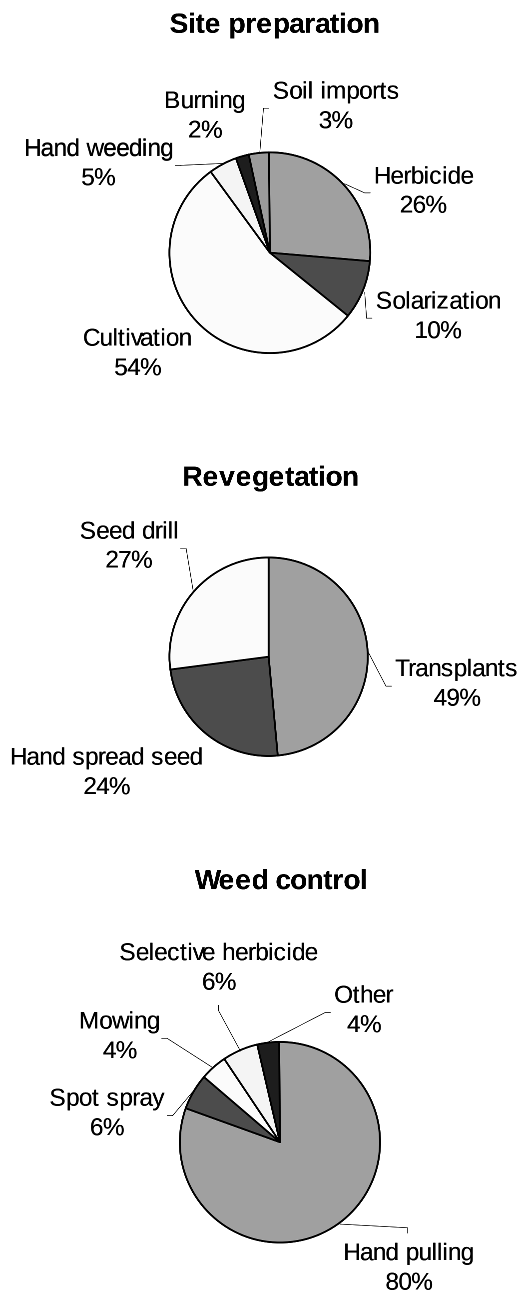


Figure 4.1. Mean relative importance (%) of management activities, as identified by restoration managers, for site preparation, revegetation and control of exotics for active restorations (n = 22).

Table 4.1. Mean species richness, effective species richness (ESR) and vegetation percent cover (± SE) for native and exotic species across reference sites, and restorations separated by restoration strategy.

	Active Restoration	Passive restoration	Reference	*p**
Species richness				
Native	4.94 ± 0.54 [b]	0.46 ± 0.15 [c]	7.76 ± 0.57 [a]	0.0008
Exotic	3.01 ± 0.24 [b]	4.33 ± 0.37 [a]	2.23 ± 0.23 [b]	0.0081
Overall	7.96 ± 0.52 [b]	4.76 ± 0.35 [c]	9.99 ± 0.64 [a]	<0.0001
Effective species richness				
Native	2.83 ± 0.29 [a]	0.36 ± 0.12 [b]	3.44 ± 0.22 [a]	<0.0001
Exotic	1.83 ± 0.12 [b]	2.33 ± 0.14 [a]	1.37 ± 0.07 [c]	0.0068
Overall	3.87 ± 0.27 [a]	2.43 ± 0.13 [b]	4.21 ± 0.26 [a]	0.0052
Vegetation percent cover				
Native	41.35 ± 1.00 [b]	1.61 ± 0.35 [c]	49.59 ± 1.23 [a]	<0.0001
Exotic	28.05 ± 1.26 [b]	78.42 ± 0.87 [a]	24.26 ± 1.17 [b]	<0.0001
Overall	69.42 ± 0.83 [c]	80.04 ± 0.82 [a]	73.89 ± 0.98 [b]	<0.0001

* *p* value of overall model statement. Means followed by the same letter lack statistical significance according to post hoc SNK test (p <0.05)

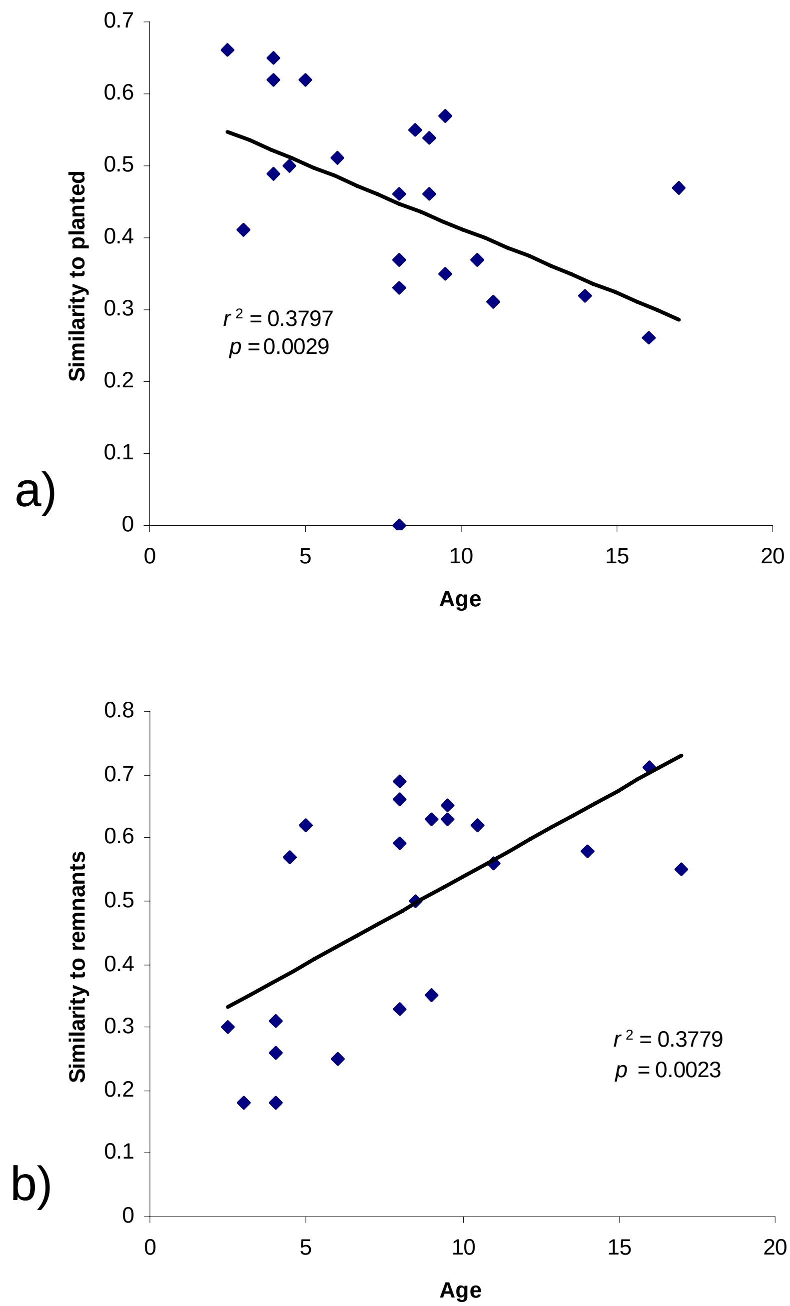


Figure 4.2. Similarity measures by age of restoration for all active restorations ($n = 22$). Indicated are (a) similarity of observed species to planted species, and (b) similarity to remnants.

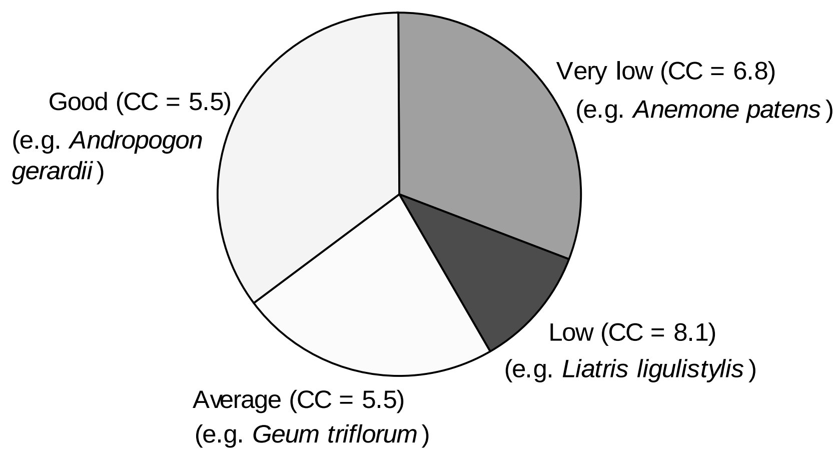
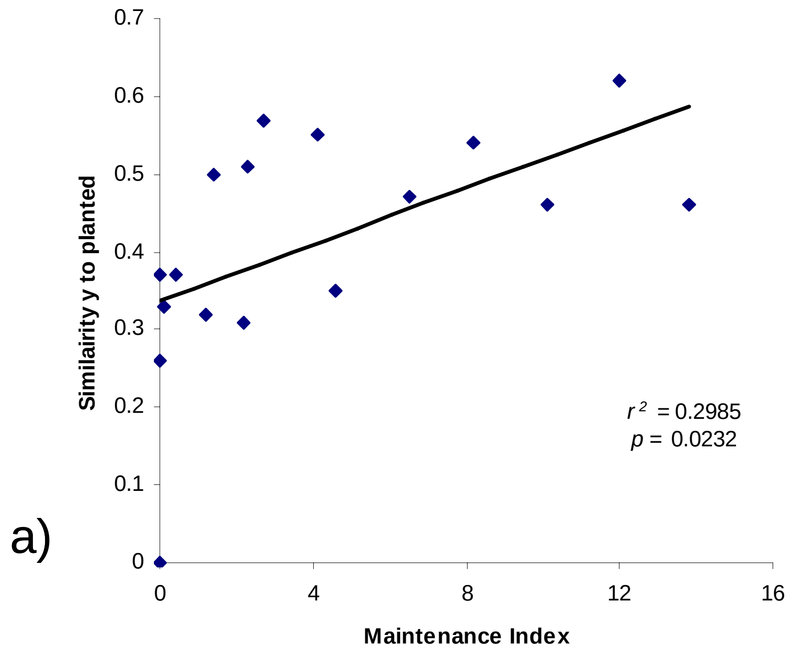


Figure 4.3. Likelihood of survival for commonly planted prairie species (n = 91) at active restoration sites (n = 22) divided into survivorship categories. Categories defined as: very low (0-24%), low (25-49%), average (50-74%) and good survival (75-100%). Mean coefficient of conservatism (CC) and typical species from each category shown.



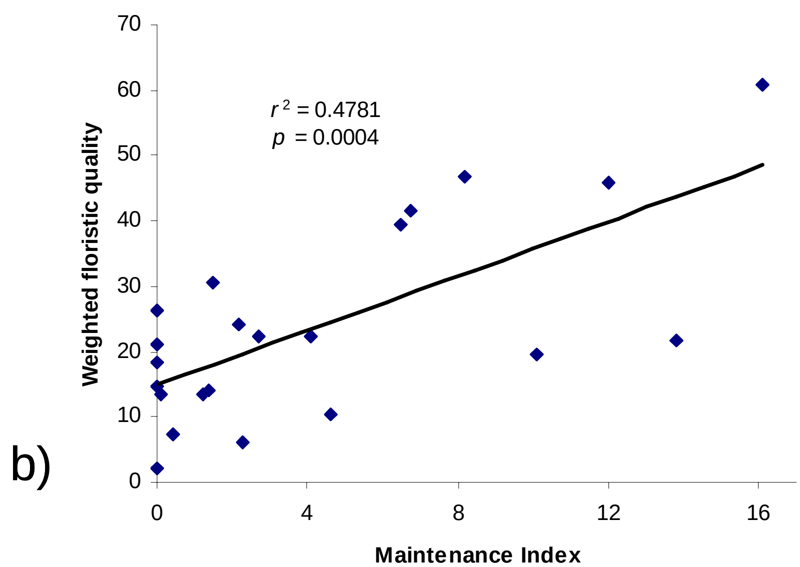


Figure 4.4. Linear regression of restoration success measures by maintenance index for all active restorations (n = 22). Indicated are: (a) similarity of observed species to planted species, and (b) weighted floristic quality.

Table 4.2. Most frequently occurring planted species and their respective coefficient of conservatism. Indicated for each is the proportion of restoration (n = 22), reference (n = 7) and nearby rural sites (n = 2) at which each species occurs.

Latin name	Common name	Proportion of restorations with species	Proportion of references with species	Presence in nearby rural prairie preserves[a]	CC[b]
Graminoid					
Andropogon gerardii	big bluestem	79.2%	71.4%	Yes	5
Panicum virgatum	switch grass	54.2%	0.0%	No	5
Sorghastrum nutans	Indian grass	45.8%	0.0%	No	6
Schizachyrium scoparium	little bluestem	41.7%	14.3%	Yes	6
Elymus canadensis	Canada wild rye	37.5%	28.6%	Yes	3
Stipa viridula	green needle grass	37.5%	14.3%	Yes	5
Bouteloua curtipendula	sideoats grama grass	29.2%	0.0%	No	8
Koeleria cristata	June grass	25.0%	0.0%	Yes	7
Agrpryron subsecundum	awned wheatgrass	25.0%	42.9%	Yes	6
Agropyron smithii	Western wheat grass	16.7%	0.0%	No	4
Agropyron trachycaulum	slender wheatgrass	16.7%	0.0%	Yes	6
Beckmannia syzigachne	slough grass	16.7%	0.0%	No	1
Deschmapsia caespitosa	hair grass	16.7%	0.0%	Yes	9
Forb					
Aster novae-angliae	new-england aster	54.2%	0.0%	No	8
Monarda fistulosa	bergamot	50.0%	0.0%	Yes	5
Aster ericoides	many-flowered aster	50.0%	100.0%	Yes	2
Galium boreale	Northern bedstraw	50.0%	85.7%	Yes	4
Heliopsis helianthoides	oxeye sunflower	41.7%	0.0%	Yes	5
Artemisia ludoviciana	prairie sage	41.7%	85.7%	Yes	3
Aster laevis	smooth aster	41.7%	85.7%	Yes	5
Helianthus maximiliani	narrow-leaved sunflower	41.7%	42.9%	Yes	5
Solidago canadensis	Canada goldenrod	41.7%	85.7%	Yes	1
Solidago rigida	stiff goldenrod	41.7%	100.0%	Yes	4
Agastache foeniculum	giant hyssop	37.5%	0.0%	No	7
Veronicastrum virginicum	Culver's root	37.5%	0.0%	No	10
Anemone canadensis	Canada anemone	37.5%	28.6%	Yes	4
Geum triflorum	three-flowered avens	37.5%	28.6%	Yes	8
Rudbeckia hirta	brown-eyed susan	37.5%	14.3%	Yes	5
Achillea millefolium	yarrow	33.3%	100.0%	Yes	3
Anaphalis margaritacea	pearly everlasting	29.2%	0.0%	No	3
Ratibida columnifera	yellow coneflower	29.2%	0.0%	No	3
Zizea aurea	golden alexander	29.2%	0.0%	Yes	8
Erigeron philadelphicus	Philadelphia fleabane	25.0%	0.0%	Yes	2
Eupatorium maculatum	joe-pye weed	25.0%	0.0%	Yes	10
Gaillardia aristata	common gaillardia	25.0%	0.0%	Yes	5

[a] (Sveinson 2003)

[b] CC, Coefficient of conservatism

Restoration strategy and ownership

Restoration outcomes were strongly affected by site ownership, significantly ($p < 0.0250$) influencing16 of the 17 measures that we examined (Table 4.3). Private restorations were most similar to the reference sites for many of the measures including native species richness and ESR for forbs and grasses (Table 4.3). However, they had more exotic forbs ($p < 0.0001$) and a higher percent cover of exotic species ($p < 0.0001$) than the reference sites. Indeed, they had higher exotic species richness than all but the passive restorations. Private/public restorations were unique in their species composition, generally having much higher native graminoid diversity than the remnants ($p < 0.0001$) but also having lower native forb diversity ($p < 0.0001$). This coupled with their low exotic graminoid diversity and cover resulted in these sites having a low similarity to the references sites (Table 4.3). In contrast, publicly owned sites that were actively restored were the most similar to the reference sites, but had relatively low native graminoid and forb diversity, the lowest native ground cover ($p < 0.0001$), and the lowest floristic quality (Table 4.3). When similarity and floristic quality were examined simultaneously, these publicly owned sites were thus most similar to the reference sites but tended to be of poor floristic quality (Fig. 4.5). On the other hand, private sites had high floristic quality but were lower in similarity to reference sites. Private/public sites tended to be of intermediate quality, although one of these was the only site in this study that had both high similarity and high floristic quality values. In contrast, passively restored sites were low for both outcome measures (Fig. 4.5).

Table 4.3. Mean (± SE) floristic variables across urban reference sites, and restorations separated by restoration strategy (i.e. active or passive) and ownership (i.e. private, private/public and public).

	Private	Priv/Pub	Public / Active	Passive	Reference	*p value* [a]
Species Richness						
Native graminoid	1.62 ± 0.11 [b]	2.76 ± 0.22 [a]	0.78 ± 0.06 [c]	0.11 ± 0.03 [d]	1.59 ± 0.07 [b]	<0.0001
Native forb	4.70 ± 0.23 [b]	3.24 ± 0.23 [c]	2.04 ± 0.17 [d]	0.31 ± 0.05 [e]	6.15 ± 0.21 [a]	<0.0001
Exotic graminoid	1.35 ± 0.08 [b]	0.98 ± 0.07 [c]	1.31 ± 0.06 [b]	2.13 ± 0.06 [a]	1.16 ± 0.04 [b]	<0.0001
Exotic forb	2.07 ± 0.12 [a]	1.44 ± 0.09 [b]	1.86 ± 0.10 [a]	2.20 ± 0.12 [a]	1.06 ± 0.08 [c]	<0.0001
Overall	9.74 ± 0.27 [b]	8.42 ± 0.22 [b]	5.98 ± 0.22 [c]	4.74 ± 0.15 [d]	9.97 ± 0.25 [a]	<0.0001
Effective species richness						
Native graminoid	1.37 ± 0.09 [b]	2.12 ± 0.16 [a]	0.71 ± 0.05 [c]	0.11 ± 0.03 [d]	1.26 ± 0.04 [b]	<0.0001
Native forb	2.54 ± 0.12 [b]	1.90 ± 0.13 [c]	1.34 ± 0.10 [d]	0.28 ± 0.05 [e]	3.34 ± 0.11 [a]	<0.0001
Exotic graminoid	1.14 ± 0.07 [b]	0.87 ± 0.06 [c]	1.10 ± 0.04 [b]	1.64 ± 0.04 [a]	1.05 ± 0.03 [b]	<0.0001
Exotic forb	1.57 ± 0.08 [a]	1.22 ± 0.08 [b]	1.35 ± 0.07 [ab]	1.49 ± 0.06 [a]	0.90 ± 0.06 [c]	<0.0001
Overall	4.61 ± 0.14 [a]	4.29 ± 0.14 [a]	2.85 ± 0.12 [b]	2.43 ± 0.07 [c]	4.20 ± 0.12 [a]	<0.0001
Percent cover						
Native graminoid	14.56 ± 1.11 [b]	29.56 ± 1.78 [a]	21.48 ± 1.94 [b]	0.40 ± 0.13 [c]	25.21 ± 1.27 [a]	<0.0001
Native forb	26.15 ± 1.43 [a]	23.43 ± 1.96 [b]	10.35 ± 1.05 [c]	1.22 ± 0.31 [d]	24.42 ± 0.96 [a]	<0.0001
Exotic graminoid	20.57 ± 1.87 [d]	11.71 ± 1.41 [e]	28.83 ± 1.82 [b]	61.23 ± 1.28 [a]	20.98 ± 1.07 [c]	<0.0001
Exotic forbs	7.70 ± 0.91 [c]	3.57 ± 0.61 [d]	10.24 ± 0.88 [b]	17.22 ± 1.28 [a]	3.31 ± 0.39 [d]	<0.0001

[a] *p*-value of overall model statement presented. Means followed by the same letter lack statistical significance according to post hoc SNK test ($p < 0.05$)

Table 4.3. Continued

	Private	Priv/Pub	Public / Active	Passive	Reference	*p value*
Similarity measures						
Similarity to planted	0.54 ± 0.03 [a]	0.48 ± 0.06 [ab]	0.34 ± 0.05 [b]	N/A	N/A	0.0250
Chao-Sorensen similarity	0.50 ± 0.05 [b]	0.42 ± 0.09 [b]	0.53 ± 0.05 [b]	0.48 ± 0.05 [b]	0.76 ± 0.03 [a] *	0.0023
Floristic quality						
Weighted floristic quality	28.16 ± 4.30 [a]	31.97 ± 6.73 [a]	12.73 ± 2.66 [b]	0.23 ± 0.27 [c]	23.94 ± 2.75 [ab]	<0.0001

* Similarity among reference sites

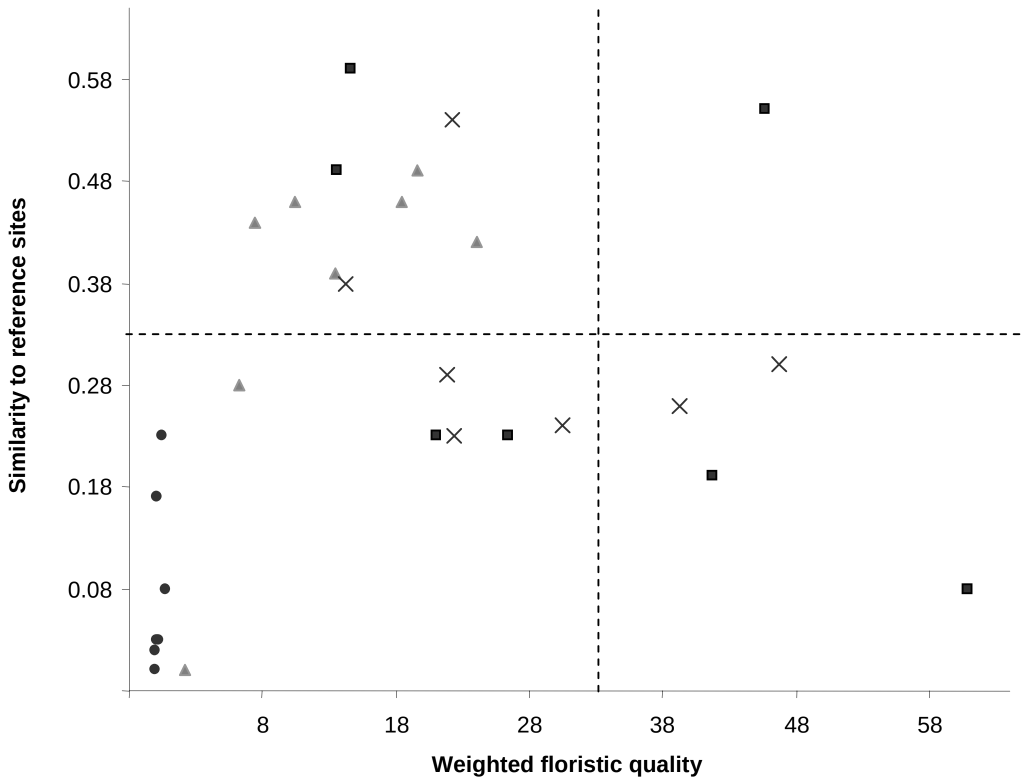


Figure 4.5. Relationship between weighted floristic quality and similarity of restorations to reference prairies for urban restorations (*n* = 29). Indicated are private restorations (X), private/public restorations (■), public restorations (▲), and passive restorations (●).

Propagule bank

The propagule bank was dominated by exotic species. Four out of the five most common propagules were exotics, the one exception, again, being *A. gerardii* (big bluestem)*,* which was the third most common seed and fourth most common rhizome (Table 4.4). Again, the most common species overall was *P. pratensis* (Kentucky bluegrass), occurring in over 90% of the sites, this followed by *Agropyron repens* (quack grass) which occurred as either rhizome or seed in 71% of the sites (Table 4.4). In total, we identified 139 species which grew from seed and 60 species from rhizomes or root buds.

There were substantial differences in species composition of rhizomes between the reference prairies and restorations. Extant prairies had the highest diversity and abundance of rhizomatous native forbs and the lowest exotic rhizome diversity (Table 4.5). However, the common occurrence of *P. pratensis* (Kentucky bluegrass) resulted in the reference prairies having a relatively high number of exotic rhizomes. This was second only to the passive restorations, which also had very few native rhizomes (Table 4.5). The active restorations differed little in rhizome diversity; although the public restorations had a lower abundance of native species than the other active restorations.

Surprisingly, the diversity and abundance of native seed was significantly higher ($p < 0.0001$, $p < 0.0001$, respectively) in the private and private/public restorations than the remnants, although they had significantly greater ($p < 0.0001$) exotic seed diversity (Table 4.5). As with rhizomes, the public restorations had significantly less ($p < 0.0001$) native seed and lower ($p < 0.0001$) seed diversity than the other active restorations. Similar to the aboveground vegetation, the passive restoration seed bank was dominated by exotics and had significantly fewer (p < 0.0001) native species (Table 4.5).

Grasshoppers and katydids

We collected 484 grasshoppers and katydids (Orthoptera), representing 5 subfamilies and 11 species (Table 4.6). The most common species was *Chorthippus curtipennis* (Meadow grasshopper) which occurred at 71% of the sites. The average density of Orthoptera was low (0.19/m^2), and, 21% of the restorations had no species at all. Diversity measures and abundance differed significantly among ownership categories and according to restoration strategy (Table 4.7). Both the private and public sites had low Orthoptera diversity and density compared to private/ public and passively restored sites (Table 4.7). The reference sites were low in density (0.09/m^2) yet had high diversity and the highest ecological integrity as reflected by the Bomar quality index. Orthoptera ESR was related to graminoid percent cover and sward height and was positively associated with graminoid cover at medium and high sward height (r^2 = 0.3879, p = 0.0407; r^2 = 0.7145, p = 0.0082 respectively); however, low numbers of sites with low and very high sward height precluded significant findings between ESR and these vegetation heights (Fig. 4.6). No relationship was found between Orthoptera diversity and plant diversity (r^2 = 0.0065).

Table 4.4. Propagule bank frequency (freq.), abundance (abund.) and rank for the five most common graminoid and forb species for both native and exotic species in restorations and remnants (n = 36).

			Rhizome		Seed		Rank by origin [a]		Overall rank [b]	
Species		Guild [c]	Freq.	Abund.	Freq.	Abund.	Rhizome	Seed	Rhizome	Seed
Exotic										
Poa pratensis	blue grass	PG	91.7%	3.28	88.9%	0.67	1	1	1	1
Agropyron repens	quackgrass	PG	63.9%	1.07	44.4%	0.07	2	4	2	7
Taraxum officinale	dandelion	PF	58.3%	0.28	52.8%	0.33	3	2	3	2
Vicia cracca	tufted vetch	PF	33.3%	0.08	5.6%	0.01	7	33	5	89
Bromus inermis	smooth brome	PG	27.8%	0.15	11.1%	0.02	5	18	6	49
Sonchus arvensis	sow thistle	PF	25.0%	0.06	36.1%	0.11	4	6	7	8
Agrostis stolonifera	redtop	PG	22.2%	0.14	19.4%	0.10	6	11	8	21
Cirsium arvense	Canada thistle	PF	11.1%	0.01	44.4%	0.20	10	5	14	5
Lotus corniculata	bird's foot trefoil	PF	5.6%	0.01	13.9%	0.09	13	14	25	29
Thlaspi arvense	penny cress	A/BF	-	-	50.0%	0.47	-	3	-	4
Chenopodium album	lamb's-quarters	A/BF	-	-	30.6%	0.33	-	7	-	9
Echinochloa crusgalli	barnyard grass	AG	-	-	16.7%	0.02	-	50	-	28
Native										
Andropogon gerardii	big bluestem	PG	55.6%	0.29	50.0%	0.12	1	1	4	3
Carex spp.	carex species	PG	19.4%	0.03	2.8%	0.00	7		9	130
Galium boreale	Northern bedstraw	PF	19.4%	0.04	16.7%	0.03	3	13	10	26

[a] Exotic and native species ranked separately, ranking of species shown for both rhizomes and seeds.

[b] All species ranked together, ranking of species shown for both rhizomes and seeds.

[c] PG = perennial grass, AG = annual grass, PF = perennial forb, AF = annual Forb, A = annual, B = biennial.

Table 4.4. Continued

Species		Guild [c]	Rhizome		Seed		Rank by origin [a]		Overall rank [b]	
			Freq.	Abund.	Freq.	Abund.	Rhizome	Seed	Rhizome	Seed
Solidago canadensis	Canada goldenrod	PF	19.4%	0.05	19.4%	0.05	2	11	11	19
Achillea millefolium	yarrow	PF	16.7%	0.04	11.1%	0.03	5	21	12	43
Aster ericoides	many-flowered aster	PF	16.7%	0.04	13.9%	0.06	6	15	13	32
Anemone canadensis	Canada anemone	PF	11.1%	0.02	-	-	9	-	15	-
Solidago rigida	stiff goldenrod	PF	8.3%	0.01	19.4%	0.03	18	12	19	20
Agropyron smithii	Western wheat grass	PG	8.3%	0.03	2.8%	0.02		57	23	102
Juncus spp.	rush	PG	2.8%	0.01	-	-	22	-		-
Potentilla norvegica	rough cinquefoil	A/B/PF	-	-	44.4%	0.19	-	2	-	6
Erigeron canadensis	Canada fleabane	AF	-	-	30.6%	0.13	-	3	-	10
Euphorbia serpyllifolia	Thyme-leaved spurge	AF	-	-	27.8%	0.13	-	4	-	12
Epilobium glandulosum	Northern willow herb	PF	-	-	25.0%	0.03	-	5	-	14
Stipa viridula	green needle grass	PG	-	-	22.2%	0.05	25	6	-	17
Agastache foeniculum	giant hyssop	PF	-	-	22.2%	0.48	23	7	-	16
Hordeum jubatum	foxtail barley	PG	-	-	11.1%	0.02	-	20	-	47
Agrpryron subsecundum	awned wheatgrass	PG	-	-	8.3%	0.01	-	27	-	59
Deschmapsia caespitosa	hair grass	PG	-	-	8.3%	0.01	-	28	-	63

[a] Exotic and native species ranked separately, ranking of species shown for both rhizomes and seeds.

[b] All species ranked together, ranking of species shown for both rhizomes and seeds.

[c] PG = perennial grass, AG = annual grass, PF = perennial forb, AF = annual Forb, A = annual, B = biennial.

Table 4.5. Mean (± SE) for propagule bank including rhizomes and seeds across urban reference sites, and restorations separated by restoration strategy and ownership in Winnipeg, Canada. *

	Private	Private/Public	Public	Passive	Remnant	*p* value*
Rhizomes						
Species richness						
Native	0.46 ± 0.08 [b]	0.54 ± 0.09 [b]	0.41 ± 0.08 [b]	0.19 ± 0.05 [c]	0.85 ± 0.12 [a]	<.00001
Exotic	0.97 ± 0.13 [b]	1.36 ± 0.15 [a]	1.24 ± 0.11 [a]	1.51 ± 0.13 [a]	1.03 ± 0.06 [ab]	<.00001
Effective species richness						
Native	1.07 ± 0.03 [b]	1.11 ± 0.04 [b]	1.01 ± 0.03 [b]	1.01 ± 0.01 [b]	1.23 ± 0.07 [a]	0.0095
Exotic	1.27 ± 0.06 [a]	1.33 ± 0.15 [a]	1.30 ± 0.06 [a]	1.38 ± 0.06 [a]	1.07 ± 0.03 [b]	0.0019
Abundance						
Native	1.17 ± 0.25 **[ab]**	0.97 ± 0.18 [ab]	0.61 ± 0.13 [b]	0.11 ± 0.04 [c]	1.42 ± 0.24 [a]	<.00001
Exotic	3.97 ± 0.60 [cd]	3.00 ± 0.53 [d]	4.69 ± 0.57 [c]	8.79 ± 0.71 [a]	5.75 ± 0.58 [b]	<.00001
Seeds						
Species richness						
Native	2.46 ± 0.27 [a]	1.20 ± 0.14 [b]	0.86 ± 0.12 [bc]	0.64 ± 0.11 [c]	1.27 ± 0.21 [b]	<.00001
Exotic	1.89 ± 0.25 [b]	1.35 ± 0.06 [bc]	1.36 ± 0.14 [bc]	2.31 ± 0.21 [a]	0.93 ± 0.10 [c]	0.0002
Effective species richness						
Native	2.01 ± 0.17 [a]	1.50 ± 0.09 [b]	1.20 ± 0.06 [c]	1.19 ± 0.07 [c]	1.44 ± 0.10 [bc]	<.00001
Exotic	1.76 ± 0.11 [b]	1.51 ± 0.09 [b]	1.52 ± 0.09 [b]	2.04 ± 0.13 [a]	1.20 ± 0.06 [c]	<.00001
Abundance						
Native	4.10 ± 0.66 [a]	4.50 ± 0.76 [a]	2.07 ± 0.40 [b]	0.37 ± 0.09 [c]	1.17 ± 0.17 [bc]	<.00001
Exotic	3.19 ± 0.65 [bc]	2.42 ± 0.52 [c]	4.02 ± 0.80 [b]	5.40 ± 0.74 [a]	2.25 ± 0.34 [bc]	<.00001

* *p*-value of overall model statement presented. Means followed by the same letter lack statistical significance according to post hoc SNK test ($p < 0.05$).

Table 4.6. Species names, densities and dietary information for grasshoppers (Acrididae) and katydids (Tettigonidae) collected from urban restorations and remnants (*n* = 24).

Family/Subfamily/species	Common name	Density* Per 100 m^2	No. sites occupied	Food preferences	Important forage families
Acrididae					
Melanoplinae					
Melanoplus bivitattus (Say)	Two-striped grasshopper	0.2	11	Polyphagous	Mustards, Legumes, Composites, Grasses
Melanoplus borealis (Fieber)	Northern grasshopper	<0.1	1	Forb feeders	Legumes, Composites
Melanoplus bruneri (Scudder)	Brunner spur-throated grasshopper	<0.1	2	Polyphagous	Legumes, Composites, Grasses
Melanoplus dawsoni (Scudder)	Dawson grasshopper	<0.1	6	Forb feeders	Legumes, Composites
Melanoplus femurrubrum (DeGeer)	Red-legged grasshopper	0.7	11	Polyphagous	Legumes, Composites, Grasses
Oedipodinae					
Dissosteira carolina (Linnaeus)	Carolina locust	<0.1	5	Polyphagous	Grasses, Composites
Gomphocerinae					
Chorthippus curtipennis (Harris)	Meadow grasshopper	1.8	17	Grass-feeders	Grasses, Sedges
Tettigoniidae					
Conocephalinae					
Conocephalus fasciatus (DeGeer)	Slender meadow katydid	1.9	16	Grass-feeders	Grasses
Conocephalus saltans (Scudder)	Prairie meadow katydid	0.1	1	Unknown	Unknown
Orchelimum gladiator (Bruner)	Gladiator meadow katydid	0.6	8	Unknown	Unknown
Phaneropterinae					
Scudderia furcata (Brunner)	Fork-tailed katydid	0.4	4	Unknown	Unknown

* Density determined from sweep net collection

Table 4.7. Mean (± SE) diversity, quality and density for grasshoppers (Acrididae) and katydids (Tettigonidae) across urban reference sites (n = 5), and restorations (n = 19) separated by restoration strategy and ownership in Winnipeg, Canada.

	Private	Private/Public	Public	Passive	Reference	p value [a]
Orthoptera diversity						
Species Richness	1.80 ± 1.11 [b]	4.75 ± 0.75 [a]	1.60 ± 1.16 [b]	4.40 ± 0.51 [a]	4.60 ± 0.51 [a]	<0.0001
Effective species richness	0.88 ± 0.54 [bc]	3.10 ± 0.58 [a]	0.64 ± 0.40 [c]	2.53 ± 0.36 [ab]	3.16 ± 0.17 [a]	<0.0001
Evenness	0.46 ± 0.27	0.63 ± 0.05	0.50 ± 0.19	0.61 ± 0.10	0.72 ± 0.07	0.7242
Bomar quality [b]	4.88 ± 3.82	13.10 ± 3.71	5.12 ± 4.32	10.72 ± 1.78	21.59 ± 6.36	0.0685
Density (Orthoptera / m^2) [c]	0.13 ± 0.05 [b]	0.20 ± 0.04 [ab]	0.11 ± 0.05 [b]	0.41 ± 0.11 [a]	0.09 ± 0.03 [b]	<0.0001

[a] p-value of overall model statement presented. Means followed by the same letter lack statistical significance according to post hoc SNK test ($p < 0.05$).

[b] Quality as defined by Bomar (2001)

[c] Density calculated from transect counts

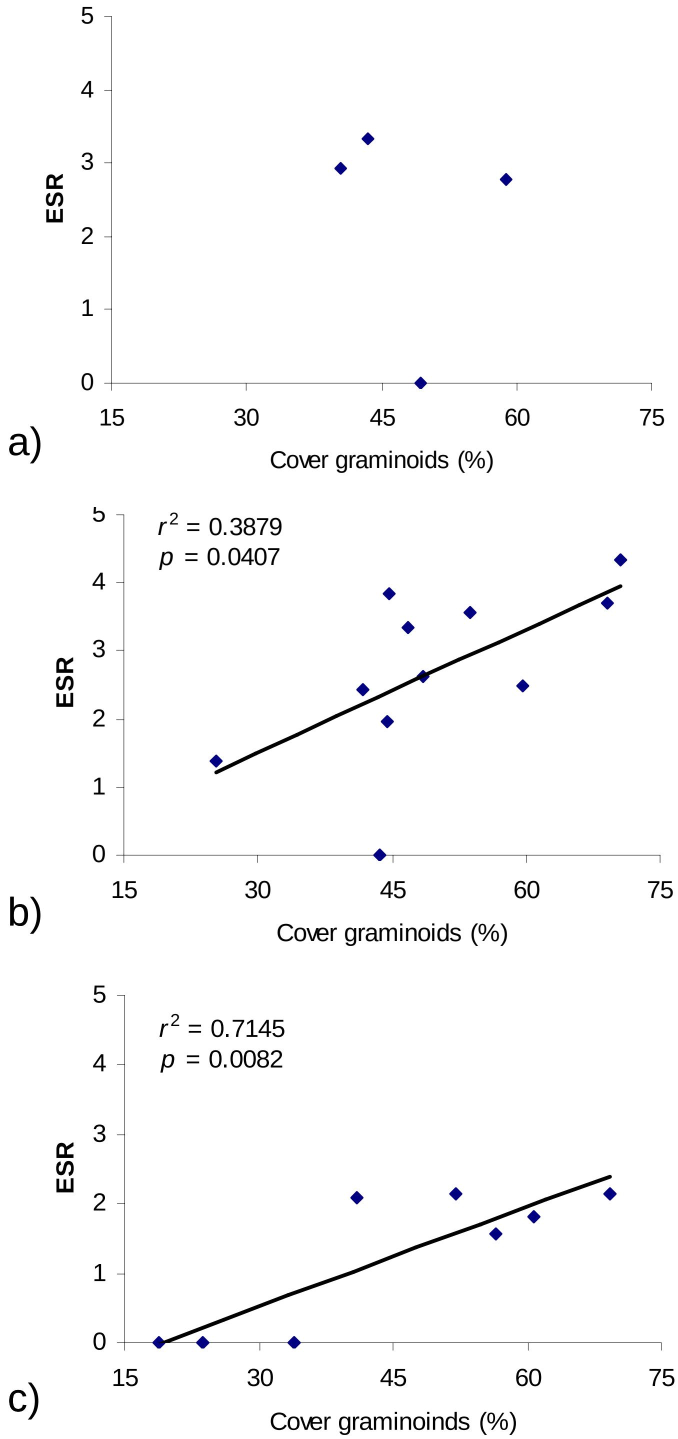


Figure 4.6. Relationship between Orthoptera effective species richness (ESR) and percent cover of graminoids at different sward heights (n = 23), where: a) low sward height, b) medium height, c) high height.

4.5 DISCUSSION

In general, the active urban restorations we examined were highly successful. In contrast, the passive restorations were notable only in their relative failure. Passive restoration has emerged as a ubiquitous approach to restoration, in part driven by low cost (De Steven 2006), and an optimistic hope that the removal of external stressors, in this case mowing, would facilitate recovery. In this study, they remained dominated by exotics and showed little if any change over time. Yet, such resistance to change is not surprising. These sites have long been abandoned, are isolated from any extant prairie habitat, and exhibit a near total absence of desirable propagules: three conditions which impose severe constraints on passive restoration (Handa & Jefferies 2000, Bossuyt & Hermy 2003, van Diggelen & Marrs 2003).

To assess the desirability of the changes associated with restoration, we used the similarity of the restored sites to neighboring extant habitat as a primary indicator of success. As with other studies (e.g., Kindscher & Tieszen 1998, McLachlan & Knispel 2005), the similarity of restorations to reference sites increased over time**.** The use of extant habitat to gauge restoration success is a widely accepted practice in the restoration literature (SER 2004). These measures have been used to gauge success in grassland (Pywell et al., 2002; Sluis 2002) and forest (McLachlan and Bazely 2001) restoration and generally increases in similarity are seen as desirable. However, the loss of desirable species over time that characterized this and many other studies (e.g., McLachlan & Bazely 2003; Sluis 2002) often results in a counter-intuitive increase in similarity. Species that disappeared in this study tended to be seed dependent forbs occurring in relatively small populations. They often were highly desirable and conservative species such as *Anemone patens* (prairie crocus). This suggests that these similarity measures are, in-of-themselves inadequate, as they are ultimately dependent on the quality of the extant habitat that is used as reference sites. It further suggests that more than one indicator of restoration success should be used.

In addition to similarity indices which are solely biological in approach, we also used floristic quality indicators which are based on criteria identified by biologists and are used to indicate the sensitivity of species to disturbance (Brudvig et al. 2007). Many active restorations actually surpassed the floristic quality of the remnants. This, in part, indicates that restoration efforts in this study tended to focus on conservative species and that conservation was the primary goal of these prairie gardeners (Kotyk 2007). Indeed, all of the commonly planted species were endemic to the region, however, some highly conservative species such as *Aster novae-angliae* (New-England aster) were absent from the nearby reference sites, and others (e.g. *V. veronicastrum* (culver's root) and *Aster sericeus* (Western silvery aster)) were listed as threatened and thus protected under provincial legislation (CCSM 1998).

Differences in success in these urban restorations can partially be explained by ownership patterns. Ownership is an important influencer of prairie management in rural areas (Higgins, Larson & Higgins 2001) and also influenced urban restorations. Both the private and private/public sites had higher diversity and lower exotic cover than public sites. This likely reflects the conservation values and priorities of the landowners, as well as the intensity of management. Rural restorations, like the public restorations in our study, are generally larger, often understaffed, and managers may be responsible for large areas (Brudvig et al. 2007). Surprisingly, even though these under-resourced public restorations had low floristic quality, as a group they were the most similar to the remnants. In contrast, a group of the more intensively managed private and private/public sites had high floristic quality and low similarity to the reference sites. Only one (a public/private site) of the 22 active restorations examined was rated as high in both "success" measures, indicating the difficulties in establishing a diverse "natural" restoration and the importance of considering multiple attributes of restorations when measuring success (Nichols & Nichols 2003, Ruiz-Jaen & Aide 2005). When the same analysis was carried out with non-weighted floristic quality and similarity measures, a much higher success rate for the restorations was achieved (11 sites ranked high with respect to both measures), demonstrating the importance of using adequately conservative measures for restoration assessments.

The mixed outcomes of these restorations indicate that urban, like rural, restoration is a long-term and labor intensive commitment. Ecosystem processes may take decades or even centuries to recover, if they return at all (McLachlan & Bazely 2001; Kucharik et al. 2006). Due to the high incidence of exotic species and habitat fragmentation of urban environments, a sustained commitment to restoration management is essential. In rural areas, prescribed burning, usually conducted in the spring, is considered to be one of the most important management activities. Although prescribed burning may inadvertently lead to the dominance of C4 grasses (Howe 1994; McLachlan & Knispel 2005), fire clears accumulated soil litter (Blair et al. 1998), modifies nutrient cycling (Blair et al. 1998), increases solar radiation to the soil (Old 1969), and helps control early season invasive species, that are ubiquitous in cities. However, the high urban population density generally precludes burning, and thus alternative small scale approaches to management such as hand weeding and thatch clearing were used. This labor-intensive approach to weed management was often facilitated by highly motivated and knowledgeable gardeners and the small size of private sites. This committed management and the use of transplanted seedlings when establishing the restorations contributed to their high floristic quality.

Restoration success commonly is assessed based solely on the diversity of vegetation (Young 2000). However, it is increasingly recognized that other components of the system should be assessed, especially those that are ancillary and not manipulated directly (Ruiz-Jaen & Aide 2005). The propagule bank is one such measure that has been identified as an important component, especially since it simultaneously reflects past management and indicates what the system might look like in the future (Johnson & Anderson 1984). Prairies are generally dominated by perennial plants (Freeman 1998); thus rhizomes are central to propagation and in some cases they account for more than 99% of total recruitment (Benson & Hartnett 2006). Although we found minimal differences in rhizome diversity among the classes of active restorations, the reference sites had more native and less exotic rhizome diversity and abundance. Yet, even these reference sites had a much higher abundance of exotic propagules than natives, largely due to the presence of *P. pratensis* (Kentucky bluegrass). Interestingly, there were more substantial differences in seed diversity among the restorations, and the more successful restorations exhibited a higher amount of native and less exotic seed. While the poor quality of the propagule banks suggests that the restorations will need to be actively managed in the future, it does indicate that these sites may eventually become more self sufficient as propagule banks develop.

The Arthropod community is another component of the tallgrass prairie that has, until recently, been overlooked as a way to assess differences in prairie quality. Hamilton used Collembola (springtails) to differentiate types of prairie, to reconstruct its former range (2005) and to indicate preserve quality (1995). Insects, including Orthoptera, have recently been used to compare restored and remnant prairies (e.g., Brand & Dunn 1998; Bomar 2001). Orthoptera may be especially useful as indicators of restoration success as they are natural dominant herbivores of the prairies, especially in urban environments, and many require particular food sources endemic to the prairie (Craig et al. 1999). Unfortunately, there is little information on their ability to colonize and establish in prairie restorations and to what degree various species are prairie specialists (Taron 1997). Like Bomar (2001), we found the Orthoptera populations on remnants were generally more diverse than the restorations and more likely to be unique species. Thus, *Conocephalus saltans* (prairie meadow katydid) and *Melanoplus dawsoni* (Dawson grasshopper), which have been identified as prairie specialists elsewhere (Reed 1996) were generally limited to high quality sites and thus have potential as indicators of restoration success. To that end, the main determinant of Orthoptera species richness was grass cover and sward height (Chambers & Samways 1998) rather than vegetation diversity as others have found (e.g., Kruess & Tscharntke 2002). On the other hand, the abundance and affinity of generalists such as *Chorthippus curtipenis* (meadow grasshopper) for sites dominated by exotics suggests that they might be useful as indictors of degradation or disturbance. Interestingly, the passive restorations had higher Orthoptera species diversity and abundance than either private or public restorations sites. This suggests that although these restorations were unsuccessful in terms of supporting native vegetation they may provide valuable habitat for Orthoptera, insects in general, and perhaps other taxa as well. This might be due to the relatively high proportion of grasses in these sites as well as their high connectivity

Implications for conservation

We have shown that restorations can be established successfully in human dominated systems, especially those that are privately owned. Private landowners arguably are the most invested in their land management and have the longest-term relationships with their restorations. As a result they are able to maintain high levels of forb diversity, particularly through small scale and intense management which is not easily achieved for large public sites. Although some argue that this 'ecological gardening' is incompatible with 'true' restoration (Throop & Purdom 2006) we feel that this intensive approach is especially important in highly disturbed urban areas as large scale sites are generally unavailable, costly and difficult to secure (Bernhardt & Palmer 2007).

Our study reinforces the important role that humans already play in urban restoration (Higgs 2003). Although our measures of success were biological, the importance of anthropogenic influences in terms of management and ownership on restoration trajectory cannot be refuted and are as, if not more, important than the biophysical factors in shaping restoration (Chapter 3). As human values and experiences are increasingly incorporated in the design and assessment of these urban restorations, it will become progressively more feasible to develop social indicators of restoration success that explicitly recognize the essential role that humans play in these activities.

While it is clear from our results that prairie restoration is compatible with urban development, the importance of protecting and managing extant habitat in cities cannot be overstated. Over the course of our research, two of the seven remnants we examined were eliminated and a third extensively damaged. That the few remaining prairie sites would be so vulnerable indicates how indifferent managers and policy makers are towards conservation, even when the ecosystems are highly endangered.

As evidenced by the large number of new urban restorations, residents are increasingly likely to value these prairies and to manage their land in ways that are compatible with prairie conservation. Indeed, threatened species such as *V. virginicum* (culver's root) and *A. sericous* (western silvery aster) seemed to thrive in urban restorations. Individually, these urban restorations represent a small increase in area for prairie conservation, but when combined together the contribution becomes significant. Thus, the combined area of the restorations in our study was over 10 ha in size. Moreover, we certainly underestimated the amount of land that contains native plants as we selected only sites that were prairies. Hybrid habitats that combine both native and cultivated plants also play a valuable role in urban conservation (Benvie pers. com.). The importance of these habitats will only increase at the landscape scale as bylaws that facilitate naturalization and eliminate pesticide use continue to be adopted across North America. In the not-too-distant future, these restorations may come to represent networks of biologically rich habitat that link people and wildlife in these cityscapes.

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Chapter 5: Thesis discussion and implications

FRAMEWORK

Rapid growth of urbanized areas has resulted in urban restoration ecology being an important field of study. Although numerous urban restorations have been carried out across North America, these restorations tend to be community based and small in scale and thus are outside the purview of much of restoration ecology. Tallgrass prairie has been identified as one of the most endangered ecosystems in North America, with less than 0.1% of its former cover remaining at the northern extent of its range, and continues to be threatened by development. Thus urban restoration of tallgrass prairie is of especial interest to conservation.

RESEARCH OUTCOMES

We found that urban prairie restorations generally were successful in supporting a diverse assemblage of native graminoids and forbs. However, there was a considerable range in quality of restorations. The most important factor influencing the success of the restorations was the strategy of restoration used. Passively restored sites were dominated by exotic species and had few, if any, desirable native species. In contrast, many active restorations were highly successful and approached the reference prairies in terms of diversity, and some even surpassed remnants in floristic quality.

Restoration studies generally focus on describing the biophysical factors influencing restoration success. However, our results showed that anthropogenic variables were at least as important in determining the outcomes of active restoration in urban environments. Seed mix and use of transplants were the two most important anthropogenic variables and had a substantial effect on the native vegetation of the sites. Ownership also explained many differences among the restorations as private restorations and private/public restorations were substantially higher than public restorations with respect to forb diversity and floristic quality. Ownership may primarily affect the sites through differences in management. Indeed, we found that management variables were also important determinants of success. A maintenance index created from management variables was found to be directly and positively related to both native forb diversity and floristic quality.

Restoration size was the most important biophysical descriptor of the vegetation; however, this finding is partially a result of changes in management regime associated with area. While larger areas are generally expected to support higher diversity, we found the opposite to be true. Native forb diversity was highest in the small restorations and actually decreased with increasing site size. This shows that managers of large sites use more grasses to simplify management and allow broad-leaved herbicide use, and further reflects the rich diversity of forbs in many of the small private restorations that resulted from small scale intensive management. Canopy cover was also found to influence the vegetation of the restorations, and sites with high levels of shade were more likely to have shade tolerant exotics such as *Arctium minus* (common burdock) and *Glechoma hederacea* (creeping ground ivy). The age of the restorations was an important biophysical descriptor and over time the restorations became more like the reference prairies. However, they also lost planted species as they aged. In general, the disappearing species tended to be conservative seed dependent forbs such as *Anemone patens* (prairie crocus), indicating the problems that can arise when using just one indicator of restoration success.

The propagule banks of all sites were dominated by exotic species and *Poa pratensis* (Kentucky bluegrass) was the most common species overall, reflecting its prevalence above ground as well. There were minimal differences in rhizome diversity among the classes of active restorations; however, reference sites had more native and less exotic rhizome diversity and abundance. Interestingly, there were more substantial differences in seed diversity among the restorations, and the more successful restorations exhibited a higher amount of native and less exotic seed. Although these sites must be actively managed in the near future, eventually they may become more self sufficient as propagule banks develop.

Grasshoper and katydid (Orthoptera) diversity was highest in the reference sites and lowest in the public restorations; however, the passive restorations had the highest density. Overall, Orthoptera diversity was not linked to the complexity of the vegetation but instead was related to the percent cover of grass and sward height. A number of species showed potential as indicators of either, prairie quality or disturbance.

Overall our results showed that urban prairies were successfully restored through active restoration and that multiple measures should be used to assess changes in quality. Indeed, the outcomes of these restorations are often more successful than their rural counterparts and have much conservation and education value. The challenge now is to increase the ubiquity of these restorations.

FUTURE DIRECTIONS

This research on terrestrial urban restorations is the first of its kind and thus provides a starting point for future studies of restoration in human dominated environments. Indeed the holistic approach used, which incorporated management variables, biophysical effects, success measures, propagule bank and insect diversity has opened up many possible future directions research could take.

As part of this study I had hoped to examine the impacts of different methods of pre-restoration site preparation; however the correlative nature of this study made this difficult. There may be considerable differences in the success of plantings depending if the pre-existing vegetation was treated with herbicides, mechanically removed or heat killed (i.e. solarization); however, it was not possible to determine what differences currently observed at the restorations resulted from site preparation and what was the result of other factors. A next step would be to set up an experiment, whereby different lengths and methods of site preparation could be compared for their success in cleaning the propagule bank and also their impacts on subsequently established vegetation.

Results from this study also suggested other future avenues of study. Passive restorations were found to be largely devoid of desirable native flora, and, as evidenced by the propagule bank, this is not expected to change. Yet, they obviously enhance biodiversity as evidenced by the diverse assemblage of grasshoppers and katydids present. Thus, a study on the potential of increasing biodiversity by re-introducing native species into these no-mow areas, possibly through interseeding or transplants and associated modifications of management to facilitate proliferation of desirables would be a valuable further study opportunity. Another prospect for research relates to the prevalence of native grasses in large restoration sites. It is evident from our results that grasslands can be established on a large scale with consistent results; however, very little is known about the subsequent re-introduction of forbs into these systems. Experimentation with methods of establishing forbs into these grass dominated systems would increase the value of these grassland restorations for wildlife as well as increasing their aesthetic appeal and, hence, their desirability and acceptance into urban environments.

Only one of the restorations examined was found to be high in both similarity to the reference prairies and floristic quality. Small intensively managed restorations were rich in forb diversity, yet generally had low abundance of grasses. Many of the managers of these small restorations expressed desire for a higher proportion of grasses, and suggested that the inability to burn their prairies in an urban environment was to blame. An examination of burn alternatives, such as mowing and raking techniques and timing that could simulate the effects of fire and control exotic species would be very beneficial for urban restoration managers.

A detailed assessment of restoration success necessitates an examination of more than just above ground vegetation. In this study I also examined management techniques, soil nutrient levels, the propagule bank and the Orthoptera community. As a result of the holistic nature of this study, all of these components were incorporated, yet none exhaustively so. Many interesting avenues remain to be explored. Of particular interest are the grasshoppers and katydids. Although there were some interesting findings, the summer of 2006 was marked by a low abundance of grasshoppers (MAFRI 2007) and thus my sample size was probably influenced. An additional year of data on Orthoptera populations where the prevalence is higher would have allowed stronger conclusions to be made. Additionally, the incorporation of larger tracts of rural remnant prairie would provide valuable baseline data about prairie specialists. Furthermore, other insects such as Leafhoppers (Cicadellidae) have shown promise as indicators of prairie quality and would be interesting to examine in the context of urban restoration.

This study begins to blur the boundaries of what is commonly referred to as *ecological restoration* and *restoration ecology;* however, there remain many opportunities for research that may further resolve this disparity. While my study has concentrated on ecological indicators of success, an honours research project by Jacqui Kotyk has examined the motivations, barriers and successes of urban prairie restorationists from a social perspective. A further study determining methods of juxtaposing these social indicators of success with ecological measures would be very intriguing, as well as providing insight on what is necessary to create culturally rich and ecologically diverse habitat in urban environments.

MANAGEMENT IMPLICATIONS

Natural areas in urban environments are highly fragmented and disturbed, yet my results indicate that urban restoration efforts have been successful at restoring areas of species rich habitat, which in many cases may exceed the diversity of rural restorations and even extant prairies. Urban prairie restoration should continue to be supported and encouraged, especially at the small scale where the results were the most successful. At the larger scale, a re-commitment to managing the prairie restorations is required and management plans ideally would be established to ensure that all sites receive attention. In situations where controlled burning is not feasible or desirable, alternate approaches such as targeted mowing should be examined to guide successional changes and control invasive species (Table 5.1).

The diversity seen at many of the reference prairies was surprising considering their relative isolation, small size and absence of management. The durability of these still high diversity remnants in a highly urbanized environment emphasizes the tremendous importance of protecting the few remaining examples of remnant prairie in Manitoba. Unfortunately, over the course of my research, two of the seven remnants examined were eliminated and a third extensively damaged suggesting current efforts for urban prairie preservation are inadequate. A concentrated effort should be made to recognize remaining prairie remnants as ecological reserves and protect them from future development. Likewise, the high level of exotics observed in the propagule banks of restorations and remnants is also a testament to the external pressure on these urban prairies and emphasizes the importance of continued management (Table 5.1). Without a continued commitment to management many of the restorations and remnants will likely become overrun with invasive species over time.

This study provides a substantial contribution to the fledgling field of urban restoration and provides a first look at restoration of the tall grass prairie in urban environments. These results have shown that high quality restorations of prairie can be carried out in urban environments by committed restorationists and we hope that the importance of these efforts will be realized. The importance of these habitats will only increase at the landscape scale as bylaws that facilitate naturalization and eliminate pesticide use continue to be adopted across North America. In the not-too-distant future, these restorations may come to represent networks of biologically rich habitat that link people and wildlife in these cityscapes.

Table 5.1. Summary with recommendations for different categories of urban prairie restoration, remnants included.

	Private	Private / Public	Public	Passive	Remnant
Strengths	• High floristic quality • Committed managers	• Species diversity • Maintenance	• Large size • Native graminoid cover	• Insect diversity • Low cost	• Resilience • High diversity of insects and plants
Challenges	• Low grass cover • Small restoration size	• Forb diversity absent from some sites • Long-term commitment	• Low forb diversity • Low maintenance • Exotic species	• Low native diversity • No source of native propagules • Invasive exotics	• Development • Disturbance
Recommendations	• Alternative manage-ment (e.g. spring mowing and raking)	• Continued commitment to restoration	• Increased maintenance • Supplemental planting	• Locate near extant prairie or in areas with existing native species • Add seed or transplants	• Protect remaining remnants

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