The Impact of *Ambrosia trifida* (giant ragweed) on Native Prairie Species in an Early Prairie Restoration Project.

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ABSTRACT

As the ecological importance of prairies is becoming more recognized, the number of prairie restoration projects is increasing worldwide. One of the major challenges in restoring any disturbed ecosystem is the successful establishment of native species at the expense of invasive species. While some weedy species are gradually replaced as other, more desired, species become established, there are invasive species that, due to their level of dominance, may out-compete native species indefinitely. The objectives of this study were 1) to quantify the impact of *Ambrosia trifida* (giant ragweed) on the plant community of a newly established prairie, 2) to assess any difference in effectiveness between management practices (cutting versus pulling) of *A. trifida*, and 3) to establish a baseline vegetation survey to be used in future evaluations and research of the prairie. In March of 2011, 30 plots (each 3 m^2) were established within the 20 acre prairie including 10 control plots [C], 10 plots where *A. trifida* was selectively cut approximately 10 cm above ground level [Rc], and 10 plots where *A. trifida* was selectively pulled [Rp]. Treatment (cutting/pulling) was applied three times (April, June, and August) during the study, and in order to compare the effectiveness of the treatments, the number of removed *A. trifida* was recorded for the first two applications. Between April and June, the number of *A. trifida* decreased by a mean of 34.9 individuals in the cut plots, and increased by a mean of 12.4 individuals in the pulled plots suggesting (albeit, not significantly) that pulling may disturb the ground and promote the germination of more seeds from the seedbed. Final biomass data collection was conducted in mid August by removing the above ground biomass of all plants excluding *A. trifida* from four subplots within each of the 30 main plots, followed by drying and weighing of all biomass. A total of 172 plants from 39 species were removed for a total biomass of 1735.10 grams. In the control plots, the mean biomass was 6.73g and the species diversity (H') was 0.037. In the treatment plots, the corresponding values were 83.39g, and 2.093. This twelve-fold difference in biomass suggests that the presence of *A. trifida* has a remarkable impact on the overall community of this newly established prairie and that the correct management of *A. trifida* could expediate the restoration process.

Keywords: Restoration, Ecology, Invasive, Prairie, Amborsia trifida, ragweed

LITERATURE REVIEW

The Prairie

There are numerous definitions of a prairie depending on the specific climate, region, and the array of plants in the community. In its basic definition, a prairie is simply a grassland characterized by a dominance of herbaceous plants, especially grasses, some shrubs, and an absence of trees. Furthermore, one can add that this ecosystem depends on certain natural forces with which it has evolved over time. While most people think of prairies as arid, rainfall can and does vary greatly. In North America, for instance, the Great Plains varies in precipitation from West to East and thus, has both shortgrass steppes as well as tallgrass prairies. The coastal prairie of Louisiana is similar in community composition to the midwestern tallgrass prairies, but there are certain differences between the two. For instance, because of the higher rainfall and the potential for the coastal prairie to turn into marshland, plants like *Panicum virgatum* (switchgrass) are more common in a coastal prairie (Allain et al. 1999). Some of the other species more common in coastal prairies are *Solidago odora* (sweet goldenrod), *Asclepias rubra* (red milkweed) and the grasses *Dichanthium tenue* (slender bluestem) and *Paspalum plicatulum* (brown-seed paspalum) (Allain et al. 1999). Native Americans and European settlers used numerous plant species of the coastal prairie for food, spices, dyes, textiles and medicines (Allain et al. 1999).

The history of the prairie

The majority of the grassland biome of North America stretches West to East from the eastern edge of the Rocky Mountains to Illinois-Indiana, and North to South from Saskatchewan, Alberta to central Mexico. This great expanse of grassland developed and was maintained due to several geological, ecological and climatic factors. The latest retreat of the North American ice sheets (approximately 10,000-12,000 BP) caused the warming of the climate as well as the leveling of topography, both factors favoring the formation of the grasslands of central North America (Axelrod, 1985). Floral and faunal fossils suggest the increasing aridity during the Miocene and Pliocene restricted the forests that once dominated the Great Plains, thereby aiding in the explosive evolution and dominance of grassland species (Axelrod, 1985). This aridity aided the grasses by means of a feedback system described by Keeley & Rundel (2005):

> *Critical elements [of the expansion of grasslands] were seasonality that sustained high biomass production part of year, followed by a dry season that greatly reduced fuel moisture, coupled with a monsoon climate that generated abundant lightning-igniting fires. As woodlands became more open from burning, the high light conditions favoured C_4 grasses over C_3 grasses, and in a feedback process, the elevated productivity of C_4 grasses increased highly combustible fuel loads that further increased fire activity.*

The essential difference between C_3 and C_4 plants is their respective photosynthesis process, and C_4 plants photosynthesize faster under high light and temperature conditions due their specific enzyme pathways. Coupled with their more efficient use of water, C_4 plants are more suited to grassland environments than their counterparts.

Another important factor in the expansion and maintenance of grasslands is the coevolution of grasses and herbivores. Throughout history there were cycles of the type (grazers, browsers), and abundance of herbivores. In North America, the diversity of ungulates, (including horses, camels, pronghorns, hogs, rhinos, and elephant-like animals) as well as rodents, peaked during the Miocene to a degree that is comparable to the savannas of contemporary Africa (Webb, 1977). This rise in diversity was partially due to the increased migration of savanna species between North and South America by way of the newly formed Isthmus (land bridge) of Panama (Webb, 1977). One of the latest of the herbivorous species to have greatly contributed to the grassland ecosystem is the bison. It is believed that bison arrived from Eurasia during the Pleistocene, and that their presence greatly altered the herbivore fauna, possibly due to their high fecundity and aggressive grazing (Stebbins, 1981). Many of the previously dominant herbivores became extinct while the bison numbers continued to increase (Stebbins, 1981). While bison may have adversely affected their herbivorous counterparts to some degree, the late Pleistocene extinction of a large number of mega fauna is believed to be attributed to climate change, as well as hunting pressures from humans (Barnosky, et al. 2004). What is known is that by the Holocene, bison became one of the few dominant herbivorous species of the North American prairie. According to the earliest estimates in the mid-late 1800's, bison may have numbered between 30 million and 60 million (Knapp et al. 1999). These vast numbers, however, would plummet to just a few thousand in a short amount of time, chiefly due to anthropogenic factors including the increased hunting pressure from Native Americans as well as European settlers (Flores, 1991), the increased conversion of grassland to agricultural land (Sampson & Knopf, 1994), and the systematic slaughter of bison ordered by the U.S. military as a tactic against Native Americans. It is clear that before their decline, the enormous number of bison had great impact on the prairies of North America.

Decimation of the North American prairie

Before European settlement and subsequent agriculture, approximately 162 million hectares (ha) of prairie covered the Great Plains (Sampson & Knopf, 1994). Due to the persistent conversion of prairies to agricultural land, it is estimated that less than 1% of this land area remains as native prairie today (Sampson & Knopf, 1994). This drastic decline is greater than the declines in any other major North American ecosystem, including old-growth forests, temperate rainforests, and bottomland hardwoods (Sampson & Knopf, 1994). Prairies of the Gulf coast have also declined dramatically in Texas and Louisiana. Of the estimated 3.6 million ha of coastal prairies before European settlement, only 26,000 ha remain in Texas, and a mere 40 ha are left in Louisiana making it one of the most endangered ecosystems of the gulf south (Allain et al. 1999). As with most ecosystems, the greatest threat to coastal prairies is habitat loss due to development. In Louisiana, most grasslands were altered for growing rice, sugarcane, forage, and grain crops; thus, the remnant prairies are only found in small patches and narrow strips along railroad tracks where farming and development were not feasible (Allain et al. 1999). Typically these small patches and railroad strips are in areas where fire is suppressed for safety reasons. Many prairie species, however, depend on fire for propagation. Periodic fires speed up decomposition, thereby returning nutrients to the soil, and fires also keep trees from becoming established in prairies, while leaving the underground structures of prairie plants intact and ready for new growth. The exclusion of trees has become especially important and challenging as the range of *Triadica sebifera* (Chinese tallow tree) keeps expanding.

Importance of coastal prairies

The ecological importance of coastal prairies is increasingly evident. In addition to the aforementioned plant species more common in coastal versus northern prairies, there are an estimated 1000 plant species that provide a vital habitat for thousands of insects, birds and mammals, including over 100 butterfly species, more than 100 species of dragonfly (including an endemic species, the prairie forceptail), and many migratory and resident birds (Allain, 1999). Furthermore, despite the diminished state and limited area of the coastal prairie, it hosts more red-tailed hawk, northern harrier, white ibis, and white-faced ibis than anywhere else within the United States; it is the home of the Attwater's prairie chicken (most endangered bird in N.A.), the Gulf coast hognosed skunk, the Cagle's map turtle (both critically imperiled), and is the exclusive wintering ground of the federally-endangered whooping crane (Allain et al. 1999). Coastal prairies are also the home for 12 species of *Asclepias spp*. (milkweed) that are a vital food source for numerous tropical butterflies including the monarch and the queen. Thus, the restoration, as well as the establishment of new prairies is becoming more popular.

Ecological Restoration

As more and more ecosystems are degraded by both natural causes and human interference, there is a growing need for the restoration of such areas. The Society for Ecological Restoration (SER) defines ecological restoration as " an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability." (2004). In some cases, the elimination of a disturbance may be sufficient to initiate recovery. For instance, the removal of a dam will immediately allow the movement of aquatic species that were once unable to migrate. In other cases, restoration may require a number of interventions (introducing native species, removing exotic species, remediating the soil) and a continued commitment to management beyond the initial phase. There has been tremendous research dedicated to understanding ecosystems in order to fine-tune our 'intentional activities' and maximize their benefits, but because of the complexity of ecosystems, there are no simple or universal answers. Instead, restoration and management have to be adaptive and specific to the given situation.

Prairie Restoration

The first coordinated effort to restore a prairie was initiated between 1936 and 1941 at the University of Wisconsin-Madison by Aldo Leopold among others (Sperry, 1983). In fact, it is believed to be the first major restoration project of any ecosystem (Mlot, 1990). Now known as Curtis Prairie, this 25 ha site has been deemed a restoration success, and provides a vital source for ongoing research in restoration and community ecology. Another noteworthy and large-scale prairie restoration project was started in 1975 at the Fermi National Accelerator Laboratory in Illinois. From an initial seeding of 3.9 ha, consecutive plantings through the years have resulted in 405 ha of restored prairie (Betz 1996). As with Curtis Prairie, the longevity of the Fermilab prairie has provided invaluable insight into aspects of restoration such as: the methods of collecting, cleaning, and sowing of seed; the timing and frequency of burns; and the development of new methods for the enrichment of plantings (Betz 1996).

There are also several ongoing regional efforts to restore coastal prairies around the western Gulf of Mexico. In 1988, the Cajun Prairie Habitat Preservation Society initiated a prairie restoration on a 4 ha lot in Eunice, Louisiana (Vidrine et al. 2001). A 1995 survey showed that 250 native prairie species were established and thriving in approximately half of the field (Vidrine et al. 1995), and by 2000, most of the early aggressive successional species (native and exotic) were absent (Vidrine et al. 2001). The Cajun Prairie in Eunice represents a successful restoration project, and provides a vital reservoir of regional seeds for future projects. Another, much larger, prairie is being restored in Evangeline Parish, Louisiana. The 165 ha Duralde Wildlife Refuge was established in 1995, and its restoration continues today with the help of several non-profit organizations.

Due to his 40+ years of experience coupled with his background in science as a professor of biology, Peter Schramm is an esteemed prairie restoration professional. As part of the proceedings in the twelfth North American Prairie Conference in 1990, Schramm submitted his illuminating paper, *Prairie Restoration: A Twenty-five Year Perspective on Establishment And Management*, which discusses the methods, challenges, and opportunities of prairie restoration. Starting with management, Schramm discusses the importance of periodic burning, stating that if a manager is in a situation (usually due to rigid safety codes) where s/he can not burn a field, s/he may as well give up because without fire, a prairie will be of poor quality and will require constant maintenance. According to Schramm, spring is the best time to burn as it leaves habitat and cover for animals throughout the winter. More importantly, a spring fire converts all the dead litter into black ash, which, in combination with sunlight, provides added warmth to the topsoil. In turn, this warmth stimulates the earlier growth of prairie species, aiding them in outcompeting exotic species. Prairies should be burned annually for the first two decades, followed by burns every three to four years. If this burn regimen is followed, Schramm is confident that any 'weedy' species will eventually give way to the prairie. In fact, what tends to become more of a problem is the prairie competing with the prairie, as both grasses and forbs are becoming established. Schramm offers a solution to this by the use of "mosaic planting" in which some areas of the plantings are loaded with a forb mix while reducing the amount of grass seeds. From site preparation, seed conditioning, and site management, to the classification of four stages of restoration succession, the detailed insight presented by Schramm is invaluable to researchers and anyone interested or involved with prairie restoration.

Despite the research, knowledge and experience behind ecological restoration, there is still much confusion and disagreement about how to accurately define restoration success (Zedler, 2007). In many cases, it is difficult to set clear goals as to when a certain project can be considered finished. Moreover, even when there are definite goals in place, monitoring and documenting the progress and quantifying the success of restoration is difficult, costly, and not well understood. Nevertheless, there is consensus that some of the most important ecosystem attributes to consider are: species diversity, community structure, ecosystem function, and the sustainability of the system (SER, 2004 and Ruiz-Jaen, 2005). In the case of a prairie, this would mean: a diverse number of plant, insect, bird, and mammal species characteristic of a prairie community; maintenance by natural forces such as fire; and the ability to be resilient and self-sustaining. In quantifying these attributes, it is vital to find and define a reference ecosystem to which they can be compared (SER, 2004). If the prairie to be restored was once a bona fide prairie, one could use historical ecological descriptions if available. If this is not possible, or if the area in question was never a prairie, one can use an existing remnant prairie for the reference ecosystem.

Beyond the scientific limitations, there are also managerial, financial, and cultural issues to consider. More often than not, restoration is not a project with a clear end in sight. There may never be a ribbon cutting. Instead, if and when the restoration phase is complete, the continuous management of the ecosystem takes over, and this transition should be seamless. Moreover, even the initial phase may take decades before differences are visibly discernable. For this first 5-10 years, a young prairie looks like a weedy, unkempt field, especially to people who are used to meticulously manicured landscapes. Additionally, most restoration projects have limited budgets and high expectations from stakeholders who want to see positive results as soon as possible. Accomplishing this requires a great deal of commitment and ingenuity.

Ambrosia trifida (giant ragweed)

As Peter Schramm has pointed out, the control of non-native weeds is one of the most unpredictable variables in restoration (1990). While *A. trifida* is not an exotic species, it is an invasive plant known as a highly dominant weed. In their 1979 study, Abul-Fatih and Bazzaz state that *A. trifida* is a fast growing annual colonizer of disturbed ground that can drastically reduce the diversity, richness and growth of other annuals. They found that the seeds of *A. trifida* germinated under a wide range of temperatures (8-41°C, with an optimum between 10-24°C), a wide range of soil moisture conditions (17-55%, with an optimum between 20-33%), and a range of sowing depth from 1-16 cm with an optimum depth of 2 cm. The depth of sowing had a direct impact on the time required for seed emergence, and the seeds that emerged first had the highest probability of survival. Conversely, the plants that emerged later had reduced plant height, weight and number of seeds per plant, compared to the earlier plants. It is important to point out, however, that according to the experimental manipulations, the delayed emergence *per se* was not the cause of the mortality, but rather the competition with individuals that emerged earlier. Compared to its associated annuals, *A. trifida* has the largest seeds and seedlings, the earliest germination and emergence and a very high photosynthetic rate, giving it a critical advantage over the other species (Abul-Fatih and Bazzaz, 1979-B).

In a separate but concurrent study, Abul-Fatih and Bazzaz assessed the dominance of *A. trifida* by removing it from some plots and leaving it in others. Throughout the growing season, it was observed that *A. trifida* outgrew all other species both in height and biomass, eventually overtopping them all and forming a closed canopy. At the end of the season, the authors found that in plots with *A. trifida*, the standing aboveground plant biomass of the other species was 43.7 g/m^2 and the species diversity (H') was 0.21. On the other hand, where *A. trifida* was continuously removed, the corresponding values were 666 g/m^2 and 1.64. Additionally, seven species that were present in the managed plots were completely excluded in the plots with *A. trifida*. The authors conclude that due to its superior ability to suppress and eliminate most associated species, *A. trifida* 'behaves' as an organizer or keystone species by controlling species composition, biomass, and diversity of the community (Abul-Fatih and Bazzaz, 1979-A)

Succession

A legitimate and often-asked question is whether it is worthwhile to control for invasive species such as *A. trifida*, or if they are simply part of natural succession, eventually to be replaced by late successional species anyways. In his paper, Schramm discusses prairie succession and proposes a thorough scheme for the four developmental stages of prairie restoration. Stage I, the Initial Downgrow Weedy Stage, lasts up to three years after planting and is characterized by the dominance of several non-native annual weeds as well as a handful of native prairie annuals such as *Rudbeckia hirta* (black-eyed Susans), and the subject of this study, *Ambrosia sp.* Schramm calls this a downgrow stage because most of the desired prairie grasses and forbs are barely noticeable above ground, since most of the growth is concentrated in their roots in order to become well established. Because of this evolutionary trait, the first several years of a new prairie looks like an unkempt, blighted field. Schramm emphasizes that the level of weed dominance in this phase is variable and unpredictable, and that it depends on the existing seed bank, as well as the success of early weed control. Stage II (2-5 yrs.) is characterized by the intense competition among the prairie species themselves as they vie for space and resources. The annual weeds are limited and less visible at this time, and by stage III (6-12 yrs.) most of them are gone entirely. The fourth and final stage, called the Long-term Adjustment Stage by Schramm, reveals much regarding staying-power of various species. Closeout of the more successional species has completed, and long-term adjustment has begun among the remaining, more mature, climax community species. In short, Schramm states that through the regular use of fire, the prairie would eventually win over the weeds. But he also states that while some prairie species are able to compete in the early successional stage, others cannot, and so, the final product is influenced by the amount of annual weed competition at the beginning of the restoration process (Schramm, 1990).

In an undisturbed habitat, where all the members of a particular ecosystem have co-evolved over time with each other and with the abiotic (climate, fire etc.) factors, human interference is not desirable. Such is not the case, however, when talking about the active establishment, conservation and/or restoration of an ecosystem. Indeed, to reach a point at which a system is self-sustainable often requires ongoing, committed intervention. As previously mentioned, such efforts can be costly and the continued funding may depend on visible signs of progress. In the case of the current study, the site is located in the middle of City Park, directly off of one of the main thoroughfares. In the front of the field, there is an informational sign with colorful pictures explaining what a native prairie 'should' look like. Yet, because of its infancy (planted in 2010) it looks more like a neglected weed-lot in need of herbicide and mowing. The public is not responsible for knowing about succession, and appreciating the 'Downgrow stage' of the prairie, and thus, there is pressure to 'tweak' the system as much as possible without compromising ecological goals for cultural ones.

For the factors already discussed, one of the main weeds in terms of ecological dominance as well as visibility in this prairie is *A. trifida*. So while it may be true that with time and the regular use of fire, this prairie will start winning over the weeds, it is anticipated that successful management of *A. trifida* will shift the ecological advantage to the desired, late-succession species, and thus, speed up the restoration process. It is hoped that through this project, the managers will have a greater understanding of the impact of *A. trifida*, and whether it is worth while to manage it, and if so, how to manage it.

Management

As discussed earlier, it appears that the key to evening out the competitive playing field is to keep *A. trifida* from germinating earlier than its surrounding species, and that this can be accomplished with the regular use of fire. The managers of the prairie are planning to burn the field every spring, but this may not always be possible, and the field may have to be mowed instead. While this is a 'next-best thing', the heavy tractor used for mowing can cause unwanted soil disturbance. Furthermore, as more of the desired prairie plants are ubiquitous in the field, it might be counterproductive to mow everything down with heavy machinery for a few patches of *A. trifida*. Instead, it may be feasible to control for *A. trifida* selectively by manually cutting and/or pulling. While there has been plenty of debate as to which method (cutting or pulling) is more effective in keeping weeds from reappearing, there does not seem to be any agreement. Mowing a field is certainly easier, while the adage "pull 'em by the roots" is accepted as scientific proof for the advantage of pulling. Cutting does not kill the whole plant, but pulling causes more unwanted soil disturbance, thereby possibly catalyzing the germination of more seeds from the seedbed. The correct method may depend on the specific plant, and the seedbed of the area in question. Therefore, another part of this study is to compare the effects of cutting versus pulling of *A. trifida* on its ability to reemerge.

INTRODUCTION

Objectives

The first objective of this study was to quantify the impact of *Ambrosia trifida* (giant ragweed) on the plant community of a newly established prairie. This was accomplished by setting up experimental plots in which *A. trifida* was either removed or left in place, and comparing the resultant biomass of plants other than *A. trifida* within these plots after a 4 month growing period. It is anticipated that this will answer an important question: In eliminating *A. trifida* as a competitor, would native prairie species be more successful, or is *A. trifida* simply utilizing space that is otherwise deficient in other species anyways? The second objective was to use two different methods of removal (cutting versus pulling) for the first 2 months in order to assess any difference in effectiveness between these methods.

This was done to gain additional insight into the ecology of *A. trifida*, and to offer quantified data to the managers of the field. Finally, this study will also result in a baseline vegetation survey that can be utilized in future evaluations of the successional process of this prairie restoration project

Site Description

The study site is a 1.75 ha field that is part of the much larger Couturie Forest and Arboretum located in City Park of New Orleans, Louisiana (Figure 1 and 4). The 14 ha preserve was established as a community arboretum in 1939 thanks to a $50,000 donation from Rene Couturie (City Park, 2011). After numerous intervals of management and neglect, the arboretum was upgraded with new interpretive trails, an amphitheater, and six education stations in 2001. The forest and surrounding waterways provide vital habitat for more than 100 species of songbirds, ducks, waders, hoot owls, white and brown pelicans, and even feral chickens. During migration season, one can also spot birds like sharp-shinned hawk, yellow-bellied sapsucker, ruby-crowned kinglet, American robin and many others. In addition to birds, one can also see alligators, box turtles, and according to some reports, even coyotes.

The devastating impact hurricane Katrina had on New Orleans and the entire northern gulf coast in the summer of 2005 cannot be overstated, and the Couturie Forest Arboretum had its fair share of destruction. Approximately 75% of the trees were severely damaged, and up to 50% were killed due to the combination of high winds and floodwaters as high as 1.8 m (Michaels, 2009). While this was not the first hurricane to affect the area, several factors contributed to cause more damage in the aftermath than in the past. In addition to wind damage, the amount and duration of floodwater during hurricane Katrina is believed to have exacerbated tree mortalities. More importantly, because of the general neglect of the area during previous storms, all fallen trees were left to decompose in the forest. After Katrina, however, most of the downed trees were removed using heavy machinery. Unfortunately, by removing trees, the nutrients that are normally recycled in natural conditions were also removed. Furthermore, the disturbed soil from heavy machinery, coupled with the opening of the canopy due to the removal of so many trees created a situation ripe for the invasion of exotic and otherwise opportunistic species (Michaels, 2009). Indeed, while there were approximately 221 exotic trees, mostly *Sapium sebiferum* (Chinese tallow) before Katrina, around 11,000 new *S. sebiferum* saplings have sprouted in patches that correspond to the removal of downed trees following Katrina (Michaels, 2009).

Restoration of Couturie Forest Arboretum

Fortunately, the managers of City Park turned the challenge of restoring the park into an opportunity to expand its natural areas (Michaels, 2009). In April 2008, Mossop+Michaels Landscape Architects were commissioned with a team of specialist sub-consultants to develop a plan that would address the strengthening of natural habitat for indigenous species of birds and animals, to develop an interpretive and educational strategy, to develop a strategy for circulation and access and to address both short and long term management of the site (Michaels, 2009). The plans will double the footprint of the natural area to approximately 28 ha split up into eight distinct ecosystems: coastal prairie, coastal marsh, eastern pine savannah bottomland hardwood, upland hardwood, live oak forest, cypress &

tupelo swamp, and riparian edge (Figure 1)(Michaels, 2009). With the help of over 3,000 volunteers, City Park has planted over 2,000 trees and removed thousands of *S. sebiferum* since hurricane Katrina.

According to the restoration plan, 8.5 ha of the 28 ha Couturie Forest Arboretum area is designated to be a coastal prairie ecosystem. The study site for this research was conducted on a 1.75 ha section of this coastal prairie situated on the South side of Harrison avenue. Marc Pastorek of Meadowmakers ™, the consultant chosen to establish and manage the prairie, conducted an initial survey of the field between November of 2008 and May 2009 to assure that the site was suitable for prairie species. Pastorek documented the state of the field prior to intervention as follows:

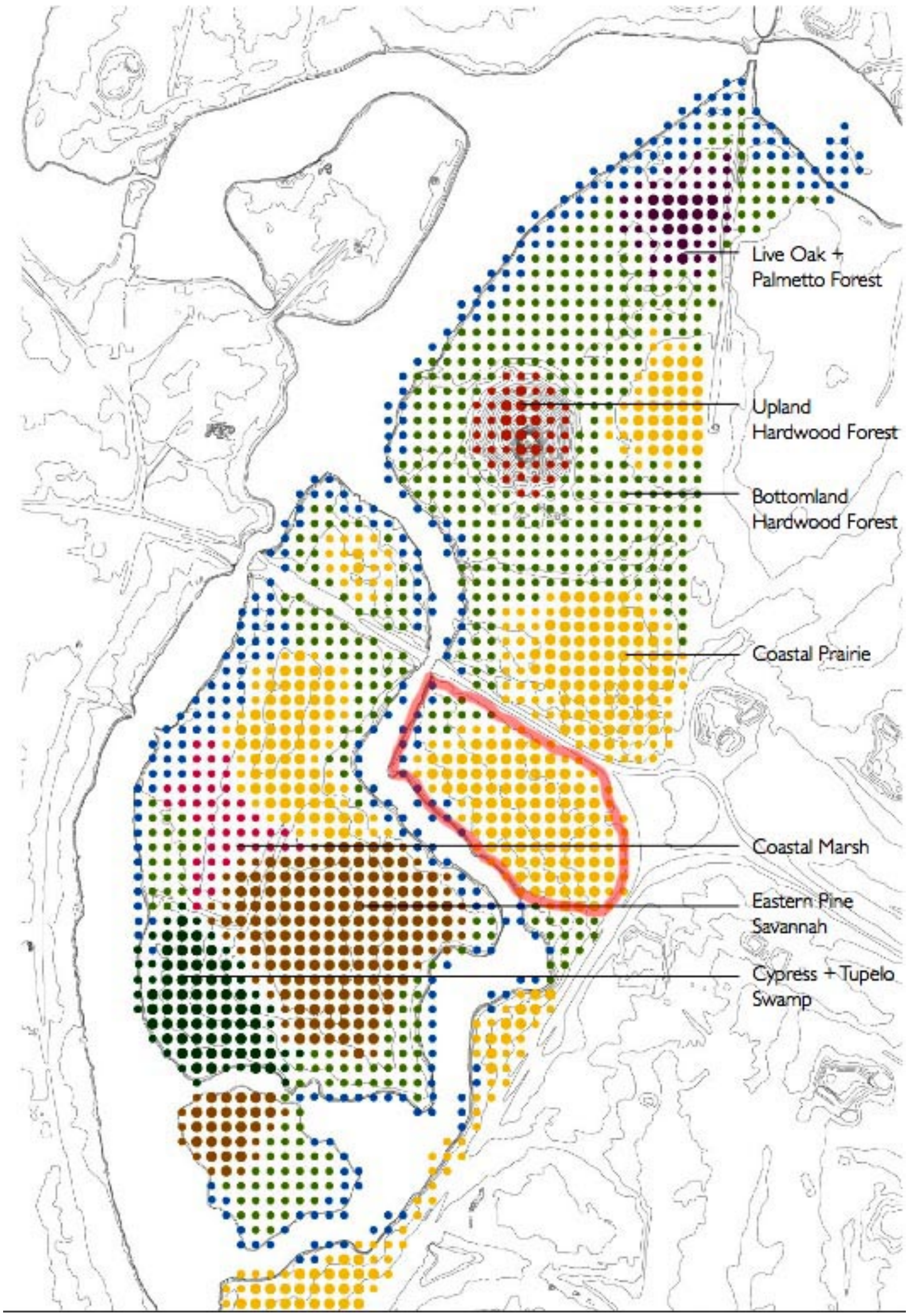
> *The landscape at Harrison Avenue was mostly dominated by Bermuda grass, mixed with a both native and introduced grasses and forbs: mostly old field weedy species. A few small trees were scattered across the landscape, but they had been mowed around for some time and had grown to ten feet or higher. Large Live Oaks edged the field with the exception of a single mature Water Oak, which happened to be in the south and central part of the field. Only five species surveyed were listed as prairie-savanna species and those were noted to be of the conservancy level of four or less. Dominant of these was Canada Wild Rye grass Elymus canadensis. The restoration plot is about four acres in size. Soil consistency is that of native marsh relic, common in the drained marshes of Orleans Parish.* [sic]

As part of site preparation, and in order to reduce the initial number of existing exotic and otherwise invasive species, two applications of non-selective herbicide were made in July and September of 2009. In November, all of the soil was turned with a disc plow to a depth of 6in to 8in. Additionally, the planting area was gridded into varying sized blocks to separate certain collections of seed for comparison in the future. Meanwhile, the seed collection for the species to be planted was taking place between April and November of 2009. All seeds were hand collected, seed-stripped or vacuumed via a flail-vac from the remaining remnants of several coastal prairies as well as from the Eunice Prairie Restoration site mentioned previously. The list of targeted species to introduce was derived from the list presented by Allen and Vidrine (2001) in their survey of the Cajun prairie of southwestern Louisiana. The final seed mix was planted between November and December 2009. In addition to the seeding, a random arrangement of nursery-grown plugs were planted to speed up the presence of hard to establish, high-target species including: *Pycnanthemum muticum* (mountain mint), *Helianthus mollis* (ashy sunflower), and *H. angustifolius* (swamp sunflower). These species are colonizing species that 'act' as companion plants for higher-conservancy species (Pastorek, 2010)

After all seeding and planting was completed, it was a matter of patience and hope to see what would happen in the first growing season of 2010. Not surprisingly, Pastorek describes the general state of the prairie as one of newness, in the early stages of succession. Although many old-field, weedy species were abundant (as expected), a surprising amount of the targeted species were present as well.

As previously discussed, however, because of the nature of 'downgrowth' most targeted species had limited aboveground presence. With time and proper management, these desired prairie species would 'hold their own' and eventually out compete the more invasive species. Pastorek and others noted that one of the main management hurdles was the suppression of *A. trifida* as it was dominating large sections of the field. Thus, the project managers were enthusiastic and supportive of this research.

<u>Figure 1.</u> The Couturie Forest and Scout Island restoration plan showing the eight proposed ecosystems. The red boundary is the site used in this study. (Image adapted from Michaels, 2009).



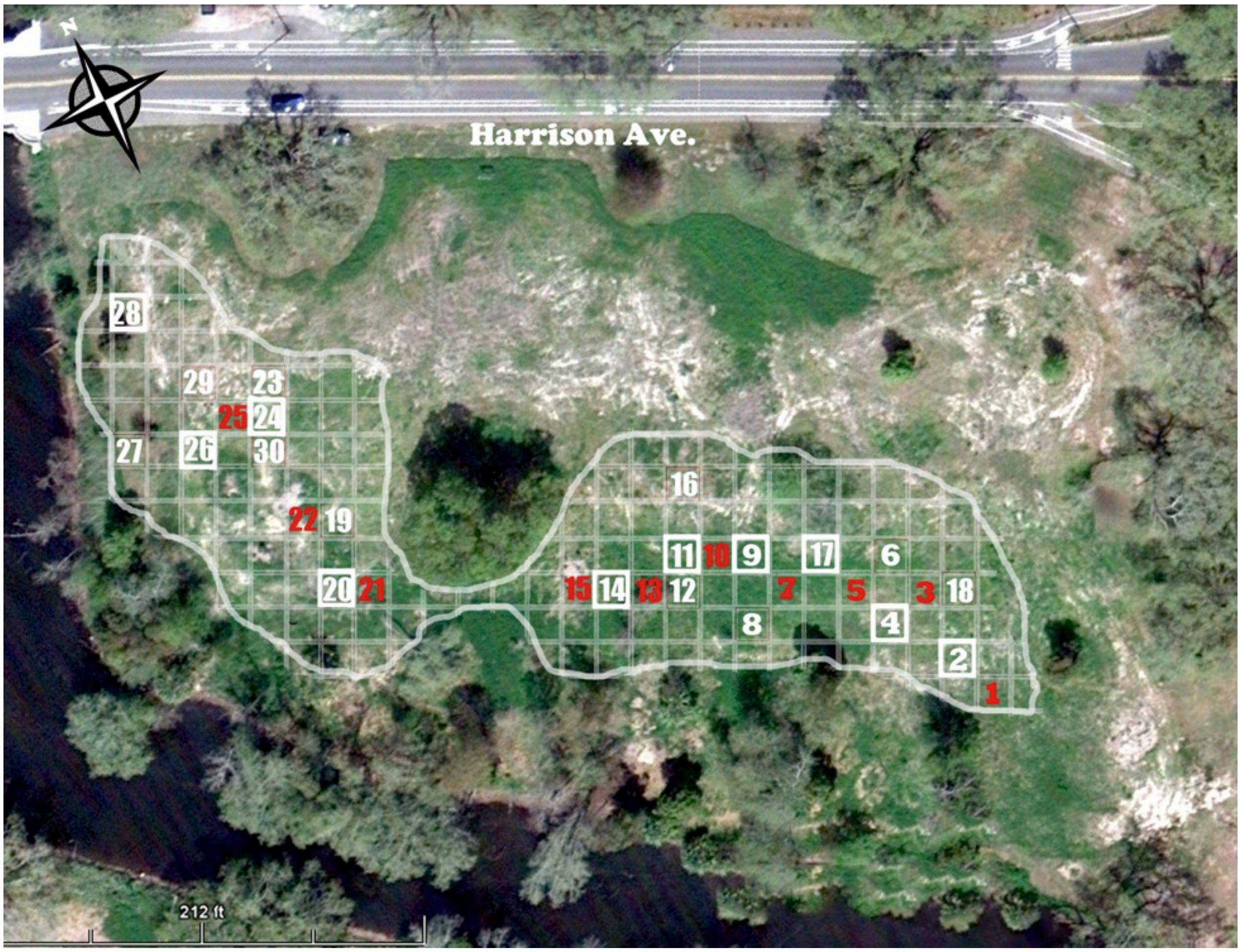
METHODS

Spring mowing of the study site was completed at the end of February 2011. Prior to setting up the experimental plots, the field was allowed to grow through March until *A. trifida* was identifiable throughout the field. This was done to assure that all study plots would be in areas where *A. trifida* was present. This wait paid off, as the majority of *A. trifida* was indeed concentrated in an East-West band on the South side of the field.

In the beginning of April, a map of the field with the study sites was created using a Google Earth™ image as the base map. First, a border was created surrounding the area where *A. trifida* was present, representing the potential study area. Next, a grid consisting of 3m squares was overlain on the map, yielding approximately 128 potential study plots. It was noted in the field that although *A. trifida* was ubiquitous within this main boundary, due to the variability in environmental factors (soil quality, hrs. of sun exposure etc.), there was some difference in the distribution. In other words, some 'clumps' of *A. trifida* seemed to be growing faster than others. Thus, while a completely random selection of study plots has its benefits, it was decided that a more systematic method might be appropriate to minimize the effects of outliers due to clumping. Consequently, the area was first divided into alternating North-South bands between plots where *A. trifida* was to be removed [R] and the control plots [C]. Then, the final location of a particular plot within its appropriate band was randomly chosen. For instance, the first control plot (plot 1) on the eastern-most boundary had 4 potential squares within the [C] band, and its final position was randomly chosen from these 4 squares. Removal plots 2 and 18 were randomly chosen from the 5 potential squares within that [R] band, and so on. As previously mentioned, part of this study was to assess the difference between cutting versus pulling *A. trifida* as a means of removal. Therefore, the [R] plots were split into two treatment plots: cut plots [Rc] and pull plots [Rp]. On the map, the [Rc] plots are indicated by a white square around the plot number. Cut versus pull plots were split up in a similar, semi-random, method to assure an even distribution around the study area. The resulting study area is composed of 10 plots each for control [C], remove by pulling [Rp], and remove by cutting [Rc] (Figure 2).

Using this finished map, the center of each field plot was designated as accurately as possible using a measuring tape, a compass, and landmarks on the map (trees, road markings etc.). Each square was marked out using a 3 m square constructed from ½" pvc pipe. The squares were then permanently demarcated with orange contractor tape and four corner flags. As a precaution to people removing any flags, a 6" metal rebar was hammered flush with the ground in the middle of each plot.

Figure 2. Location of the study plots. Within the treatment sites, the plain white numbers are pull (Rm) plots, the boxed sites are cut [Rc] plots, while the Control [C] plots are denoted in red.



Baseline Data

Following the establishment of the study plots, baseline data were taken of the existing cover percentage of *A. trifida* in each plot. In order to be as accurate and consistent, as possible, a cover estimation method described in Elzinga et al. (2000) was employed. A 50 cm square frame with a handle was constructed out of pvc and divided via string into four 25 cm squares (Figure 5). As the study plots were 3 m^2, the entire 50 cm^2 frame and the smaller 25 cm squares represented 2.78% and 0.69% cover percentages, respectively. For instance, a count of 9 whole 50 cm^2 frames would equal 25% cover.

Removal (Cutting and/or Pulling)

The first period of *A. trifida* removal occurred on April 15th-17th. In the ten [Rc] plots, all *A. trifida* over 30 cm were cut at ground level, and in the ten [Rp] plots all *A. trifida* over 30 cm were pulled (including roots). Control plots [C] received no treatment. The number of removed *A. trifida* was recorded for later comparison.

The second period of removal occurred approximately 2 months later, between June 12th-14th, following the same procedure as above.

The third period of removal occurred on August 12th, but as previously explained, only the first two treatment periods were to be used to assess the difference between pulling and cutting, so in this last period *A. trifida* was removed from all [R] plots using a weed eater. Again, the control plots [C] received no treatment.

Final data collection was conducted on August 14th and 15th using the following procedure for all 30 study plots. Four sampling plots (constructed of 50 cm^2 pvc frames) were placed within the 3 m^2 study plot yielding a total sampling area of 1 m^2. Sampling plots were positioned in the same place in all 30 study plots as Figure 6 illustrates. To place the 50 cm^2 frames in the *A. trifida* dominated [C] plots, the frames had to be partially disassembled, the two halves set in their place, and then reassembled. Once all four frames were in place, all species (other than *A. trifida*) whose stems were within the frame boundary were cut flush with the ground (in the case of vines, only the section of the vine residing within the frame was removed). Then, with the help of two botanists, each species was placed in a separate brown paper bag, all of which were then placed inside one plastic contractor bag for each study plot.

On August 16th all collected samples were transported to a lab at the University of New Orleans. Left in their brown paper bags, half of all the samples fit into three separate drying ovens, and were left drying at 75°C for 48 hours. To assure that samples were consistently dried, samples from all three ovens were periodically removed, weighed, and returned to the ovens. After the 48 hours, the first batch of samples was removed, and each bag was weighed using a digital scale. The second batch of samples was dried and weighed following the same procedure.

In order to conduct some of the data analysis (MDS, ANOSIM, Shannon diversity index), the [C] plots required biomass data for *A. trifida*. However, since *A.*

trifida was not removed and weighed, the biomass had to be estimated. While it is always better to use observed numbers instead of estimates, the large amount of *A. trifida* within the study area would have been overwhelming to remove, dry and weigh. Thus, during sampling, the number of *A. trifida* stalks within the sampling quadrats was counted for each [C] plot. These numbers were used in conjunction with a study by Abul-Fatih and Bazzaz, in which the final standing biomass of *A. trifida* is correlated to the density (1979-C). To assure a conservative estimate, the lowest values were used in all ranges involved.

Collected data were analyzed with PRIMER v5 software (Clarke and Warwick, 2001). Multidimensional scaling (MDS) ordination plots were constructed to visualize separation between [C], [Rc], and [Rp] sites. The distance between any two samples on an MDS plot correlate to the relative similarity between them. To support the MDS ordinations with statistical values, analysis of similarity tests (ANOSIM) were conducted yielding global R-values and P-values. R-values range from 0 (indicating that no significant difference exist between treatments) to 1 (indicating that all sites within a particular treatment group are more similar to each other than any other sites from different treatment groups)(Clarke and Warwick, 2001). P-values smaller than 0.05 indicate that the found values are significant. Species diversity was calculated using a version of the Shannon diversity index:

$$\overline{H} = -\sum_{i=1}^{n} p_i \log_2 p_i,$$

where n is the number of species and p_i is the proportional biomass of each species.

RESULTS

Baseline Data

Estimates of the average initial cover percentage of *A. trifida* per plot were 23% for [C], 20% for [Rp], and 25% for [Rc] (Table 1) indicating an even distribution across all study sites.

Removal (Cutting versus Pulling)

For [Rp] plots, the number of *A. trifida* pulled was 2,067 in April and 2,191 in June, yielding a total increase of 124 individuals between removal intervals with a mean increase of 12.4±80.60, or 5.99% per plot. For [Rc] plots, the number of *A. trifida* cut decreased by 349 individuals from 2,547 in April to 2,198 in June for a mean decrease of -34.9±42.6, or 13.70% per plot.

Table 1. Results showing initial cover estimates for all plots. For the removal plots, the number of *A. trifida* cut and pulled in April and June are also shown. Δx represents the mean change of removed *A. trifida* per plot.

	PLOT	Cover	(R) Apr	(R) Jun	Δ Jun-Apr
REMOVE CUT	02	19%	211	150	-61
	04	39%	458	440	-18
	09	31%	287	148	-139
	11	33%	295	304	9
	14	33%	330	311	-19
	17	17%	150	118	-32
	20	25%	403	390	-13
	24	19%	147	135	-12
	26	17%	186	128	-58
	28	14%	80	74	-6
	Σ	**-**	**2547**	**2198**	**-349**
	x̄	**25%**	**254.7**	**219.8**	**-34.9**
	SD		**±120.8**	**±129.2**	**±42.6**
					Δx̄ = -13.70%

	PLOT	Cover	(R) Apr	(R) Jun	Δ Jun-Apr
REMOVE PULL	06	14%	151	66	-85
	08	11%	115	170	55
	12	35%	220	275	55
	16	25%	205	396	191
	18	19%	180	95	-85
	19	31%	321	312	-9
	23	17%	124	131	7
	27	14%	288	243	-45
	29	19%	187	201	14
	30	19%	276	302	26
	Σ	**-**	**2067**	**2191**	**124**
	x̄	**20%**	**206.7**	**219.1**	**12.4**
	SD		**±70.0**	**±105.3**	**±80.6**
					Δx̄ = 5.99%

	PLOT	Cover
CONTROL	01	13%
	03	11%
	05	18%
	07	19%
	10	47%
	13	15%
	15	47%
	21	25%
	22	14%
	25	19%
	x̄	**23%**

Species Survey Data

A total of 38 species were collected including 20 forbs, 10 grasses, 4 trees (all seedlings) and 4 vines. The top three species with the highest relative frequency (number of plots inhabited) were *Sida rhombifolia* (12.05%), *Quercus virginiana* (9.64%), and *Trifolium L.* (7.2 3%). The top three species with the highest relative biomass were *Sida rhombifolia* (37.45%), *Symphyotrichum pilosum* (19.01%), and *Solidago canadensis* (13.71%). As previously described, these preceding values were calculated to yield a final importance value on a scale of 1-100 for each species. The top five species in order of importance were *Sida rhombifolia* (24.75), *Symphyotrichum pilosum* (10.11), *Solidago canadensis* (7.46), *Quercus virginiana* (5.46), and *Rubus spp*. (5.41). Out of the 38 collected species, only four are considered introduced, or exotic, and nine species were intentionally planted as part of the prairie restoration effort. All preceding values for all species can be found in Table 2.

Table 2. Survey data of all species (excluding *A. trifida*) collected in all sites ranked by an importance value calculated from the relative biomass and relative frequency values. Species status is indicated either by N=native, I=introduced, or --- =unknown. Species that were planted as part of the restoration effort are also shown.

Species	Imprt.	Rel. Biomass	Rel. Freq	Freq.	Status	Planted
Sida rhombifolia	24.75	37.45%	12.05%	20	N	
Symphyotrichum pilosum	10.11	19.01%	1.20%	2	N	X
Solidago canadensis	7.46	13.71%	1.20%	2	N	
Quercus virginiana	5.46	1.27%	9.64%	16	N	
Rubus spp.	5.41	4.18%	6.63%	11	N	
Trifolium L.	4.25	1.27%	7.23%	12	I	
Rudbeckia maxima	3.90	7.19%	0.60%	1	N	X
Poaceae	3.50	1.57%	5.42%	9	---	
Pycnanthemum muticum	3.43	2.64%	4.22%	7	N	X
Carex spp	3.42	0.81%	6.02%	10	---	
Phyllanthus urinaria	3.24	0.45%	6.02%	10	N	
Calyptocarpus vialis	2.85	1.47%	4.22%	7	N	
Oxalis spp	2.65	0.47%	4.82%	8	---	
Cyperus spp	1.67	0.93%	2.41%	4	---	
Cynodon dactylon	1.67	0.92%	2.41%	4	I	
Digitaria haller	1.63	1.44%	1.81%	3	---	
Salvia lyrata	1.61	0.20%	3.01%	5	N	X
Ipomoea	1.59	0.16%	3.01%	5	N	
Alternanthera philoxeroides	1.27	0.73%	1.81%	3	I	
Diodia virginiana	1.21	0.61%	1.81%	3	N	
Campsis radicans	1.12	0.43%	1.81%	3	N	
Eupatorium serotinum	0.85	1.10%	0.60%	1	N	
Fraxinus pennsylvanica	0.79	0.37%	1.20%	2	N	
Commelina communis	0.62	0.03%	1.20%	2	N	
Ruellia noctiflora	0.62	0.03%	1.20%	2	N	X
Eupatorium capillifolium	0.61	0.01%	1.20%	2	N	
Sorghum halepense	0.52	0.44%	0.60%	1	I	
Celtis laevigata	0.45	0.30%	0.60%	1	N	
Paspalum spp	0.43	0.26%	0.60%	1	N	
Helianthus mollis	0.37	0.14%	0.60%	1	N	X
Coreopsis lanceolata	0.37	0.13%	0.60%	1	N	X
Cocculus carolinus	0.35	0.09%	0.60%	1	N	
Alopecurus spp	0.34	0.07%	0.60%	1	N	X
Schizachyrium nees	0.33	0.05%	0.60%	1	N	X
Scirpus spp	0.33	0.05%	0.60%	1	---	
Quercus nigra	0.31	0.02%	0.60%	1	N	
Dichondra carolinensis	0.31	0.01%	0.60%	1	N	
Rumex crispus	0.31	0.01%	0.60%	1	N	
Σ	100.00	100.00%	100.00%	166		

Main Biomass Results (Control versus Remove)

A total of 17 species were sampled from the [C] sites, with a mean of 3.3±2.26 species collected per site. At the [R] sites, there were a total of 28 species, with a mean of 6.6±3.38 species per site. The total biomass of species excluding *A. trifida* at [C] sites was 67.3 g with a mean of 6.73 ±4.74 g per site. Total biomass at all [R] sites was 1667.8 g for a mean of 83.39±117.01 g per site. More specifically within the removal sites, biomass numbers at [Rc] versus [Rp] sites were 667.7 g with a mean of 66.77±46.08 g per site, and 1,000.1 g with a mean of 100.01±91.31 g per site, respectively.

Including the *A. trifida* estimates changes these numbers dramatically. The number of *A. trifida* stalks found in the [C] plots ranged from 82 to 128 individuals per m^2. In the Abul-Fatih study, a density of 90 plants per m^2 yielded a final standing biomass of 2,154 g (1979-C). In order to obtain conservative estimates, I used 1,500 g of *A. trifida* biomass per m^2. Even so, the total biomass of *A. trifida* in all 10[C] plots becomes 15,000 g. For the analyses on MDS, ANOSIM, and biomass species diversity (H'), the estimated *A. trifida* biomass value was included. Because of this overwhelming dominance of *A. trifida*, the Shannon biomass diversity index result is 0.037 for [C] plots. In the [R] plots, on the other hand, an index of 2.093 indicates high species diversity. All biomass data is summarized in Table 3.

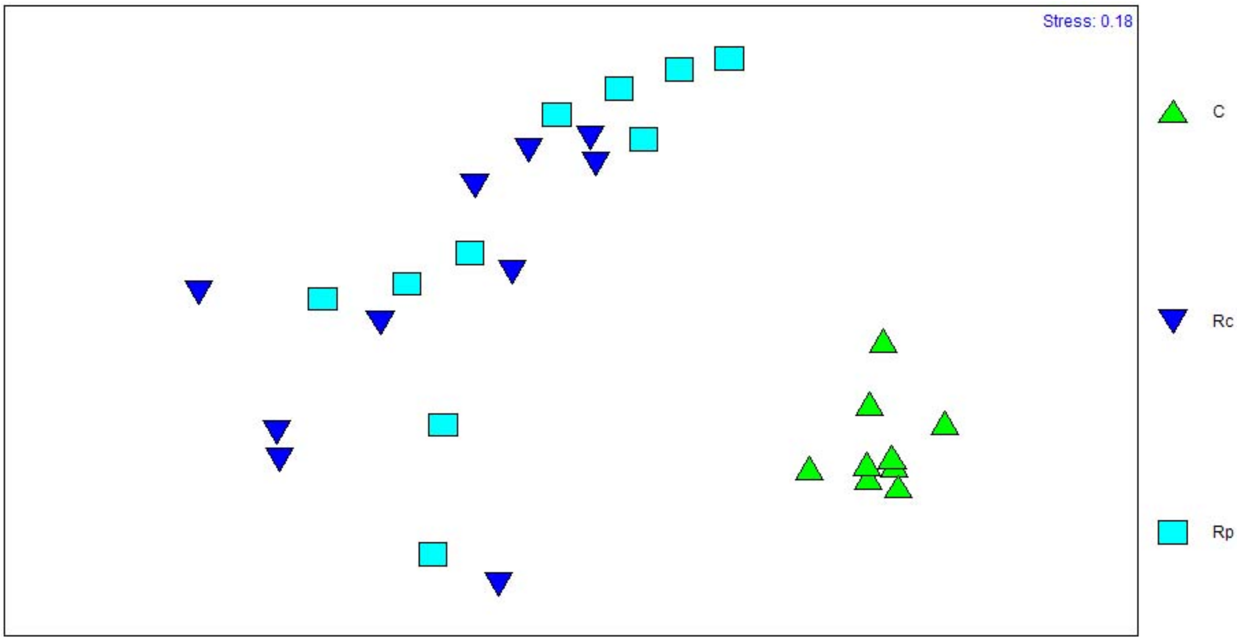
The MDS plot shown in Figure 3 shows clear separation between [C] and [R] plots, and indeed, the ANOSIM yielded R-value of 0.791 and P-value of 0.001

indicates a significant difference between the two. The differences between [Rc] and [Rp] plots, however are not significant, as indicated by a P-value of 0.94.

<u>Table 3.</u> Total biomass data in grams for all 30 plots divided into 10 [C] plots, and the 20 [R] plots with 10 plots each in [Rc] and [Rp]. Number of species, total biomass and mean biomass per site is also shown for each treatment. * Species diversity index was calculated using an estimate of 15,000 g biomass of *A. trifida* in the [C] plots.

Species	Control	Remove	Removal		Total
			Rc	Rp	
Alopecurus spp	---	1.2	1.2	---	1.2
Alternanthera philoxeroides	---	12.7	0.2	12.5	12.7
Calyptocarpus vialis	4.5	21	12.5	8.5	25.5
Campsis radicans	1	6.53	0.8	5.73	7.53
Carex spp	0.7	13.3	2.7	10.6	14
Celtis laevigata	5.2	---	---	---	5.2
Cocculus carolinus	---	1.6	---	1.6	1.6
Commelina communis	---	0.5	---	0.5	0.5
Coreopsis lanceolata	---	2.3	2.3	---	2.3
Cynodon dactylon	0.3	15.6	8.1	7.5	15.9
Cyperus spp	0.3	15.9	1	14.9	16.2
Dichondra carolinensis	---	0.1	0.1	---	0.1
Digitaria haller	---	24.9	1.1	23.8	24.9
Diodia virginiana	---	10.6	8.7	1.9	10.6
Eupatorium capillifolium	---	0.2	---	0.2	0.2
Eupatorium serotinum	---	19	19	---	19
Fraxinus pennsylvanica	3.3	3.2	3.2	---	6.5
Helianthus mollis	---	2.5	2.5	---	2.5
Ipomoea	0.1	2.6	0.4	2.2	2.7
Oxalis spp	0.2	8	3.4	4.6	8.2
Paspalum spp	---	4.5	---	4.5	4.5
Phyllanthus urinaria	---	7.8	1.8	6	7.8
Poaceae	4.4	22.8	13.1	9.7	27.2
Pycnanthemum muticum	3.5	42.3	15.4	26.9	45.8
Quercus nigra	0.3	---	---	---	0.3
Quercus virginiana	1.9	20.1	2.8	17.3	22
Rubus spp.	14	58.5	7.2	51.3	72.5
Rudbeckia maxima	---	124.8	124.8	---	124.8
Ruellia noctiflora	---	0.6	0.6	---	0.6
Rumex crispus	---	0.1	---	0.1	0.1
Salvia lyrata	0.4	3.1	1.8	1.3	3.5
Schizachyrium nees	---	0.8	---	0.8	0.8
Scirpus spp	---	0.8	---	0.8	0.8
Sida rhombifolia	25.6	624.2	120.1	504.1	649.8
Solidago canadensis	---	237.9	219.6	18.3	237.9
Sorghum halepense	---	7.6	---	7.6	7.6
Symphyotrichum pilosum	---	329.77	85.1	244.67	329.77
Trifolium L.	1.6	20.4	8.2	12.2	22
Σ of species	17	36	28	28	38
Σ Biomass in grams	67.3	1667.8	667.7	1000.1	1735.1
x̄ Biomass per site	6.73	83.39	66.77	100.01	86.76
STDV	±4.74	±117.01	±46.08	±91.31	±120.36
$\overline{H}$ **(species diversity) ***	0.037	2.093			

Figure 3. MDS plot showing the (dis)similarity within and between treatments. [Rc] and [Rp] plots are interspersed indicating no significant (P=0.94) differences. Both [R] plots however are clearly separated from the [C] plots indicating significant difference does exist (P=0.001).



DISCUSSION

Baseline Data

Since the key objective of this study was to quantify the impact of *A. trifida* on its surrounding plant community, it was important to assure that the initial cover of *A. trifida* was evenly distributed in the study plots. It must be noted that such visual estimates are subject to observer bias and inconsistency, and can vary up to 25% between observers (Elzinga et al. 2000). As described earlier, cover was estimated using 50 cm^2 quadrats in order to minimize such error in this study, and the differences between cover percentages [23% for [C], 20% for [Rp], and 25% for [Rc]], are well within a range that can be considered even.

Removal (Cutting versus Pulling)

Results show that while there was an increase of 124 *A. trifida* pulled in the [Rp] plots from April to June, the number of *A. trifida* cut in the [Rc] plots decreased by 349 individuals in the same time period. This may suggest that pulling *A. trifida*, or any plant for that matter, is not only un-advantageous, but may in fact promote the germination of even more weeds due to the soil disturbance involved. However, while a decrease of 349 plants in the [Rc] plots seems significant, it is only a 13.70% decrease per plot. What these results do show is that there is no significant advantage to pulling *A. trifida*, and since pulling is certainly much more labor intensive that cutting (mowing), it can be disregarded as a feasible management method. On the other hand, mowing with heavy machinery can, and will cause similar soil disturbance. Thus, I would suggest manual cutting of *A. trifida*, using a weed-eater for younger and mature plants, especially before flowering. A weed-eater equipped with a metal blade would even cut through the thick stalks of mature *A. trifida*. While this may seem a daunting task, if the area of *A. trifida* is limited to smaller patches, it can be done with a few volunteers in a matter of hours without causing much soil disturbance.

Species Survey Data

In the absence of *A. trifida*, the prairie is already showing a high level of species diversity (H'=2.093) without any single species dominating the area. Although *S. rhombifolia* was the most dominant species, its presence did not seem to impede the success of surrounding species. The second and third most important species, *S. pilosum* and *S. canadensis*, had high biomass values, but both were only present in two plots. Conversely, while the fourth important species, *Q. virginiana,* was present in 16 plots, all samples were small seedlings, which, due to the treeless characteristic of a prairie, will not become mature trees. It is encouraging that out of the 38 collected species, only four are considered introduced, or exotic, and that nine of the species were from the intentional planting mix used as part of the prairie restoration effort.

Main Biomass Results (Control versus Remove)

The main biomass results showed a significant difference in both species diversity and species biomass between control plots [C], and removal plots [R]. For instance, the mean number of species per plot was double in [R] plots than in [C] plots (6.6 versus 3.3). The mean biomass was only 6.73 g per [C] plot, whereas the [R] plots had a mean biomass of 83.39 g. This is more than a 12-fold increase, and it demonstrates the ability of *A. trifida* to suppress its surrounding plant community. The difference between [Rc] and [Rp] plots were less significant at 66.77 g and 100.01 g per site, respectively. The fact that the [Rp] plots had greater biomass may reinforce the earlier observation that pulling plants by the roots stimulates the germination of more seeds in the seedbed. That said, the statistical difference between [Rc] and [Rp] plots was not significant (P=0.94).

The MDS ordination plot clearly illustrates that [C] and [R] plots are significantly different. While the [Rc], and [Rp] sites are clumped together indicating relative similarity, they are both separated from the [C] sites by a large gap. This visual representation of the difference is supported by the ANOSIM yielded R-value of 0.791 and P-value of 0.001. Additionally, the Shannon biomass diversity index result was 0.037 for [C] plots and 2.093 for [R] plots. These results concur with the findings of Abul-Fatih and Bazzaz and reiterate their conclusion that *A. trifida,* due to its ability to suppress and eliminate most associated species, 'behaves' as an organizer or keystone species by controlling species composition, biomass, and diversity of the community (1979-A).

While quantitative results are the backbone to any ecological study, qualitative descriptions can also offer vital insight. From my observations of the prairie throughout the growing season, it is difficult to accept that *A. trifida* has any positive contribution to the plant community. In all areas of the study site where *A. trifida* was absent, there appeared to be a thriving community of diverse prairie plants. Most, if not all species were growing to maturity and displaying an array of colorful flowers. No other species was so dominant as to suppress its neighbors, or to take over any extensive part of the field. On the other hand, the areas where *A. trifida* was present, virtually no other plants could be easily seen. By maturity the dense stands of *A. trifida* would reach over 10 ft, allowing little sunlight to reach the ground. Any species that did manage to emerge were almost always small seedlings that would probably never reach flowering stage. In short, the data, along with the qualitative observations lead me to reject the idea that *A. trifida* is simply taking advantage of areas that are deficient in other species anyways. Instead, it is aggressively suppressing an otherwise young, but diverse prairie community in need of all available resources. And while it may be true that eventually the prairie would outcompete such invasive species, the proper management of *A. trifida* may aid in speeding up the successional process by giving other species a more level playing field.

As discussed, the best management of *A. trifida* includes several key methods. The use of regular spring burns will encourage the earlier emergence of prairie species and give them a competitive advantage (Schramm, 1992). Additionally, later in the growing season, patches of *A. trifida* should be mowed or, if possible, selectively removed using weed-eaters to minimize soil disturbance.

CONLCUSION

The prairie restoration project at City Park of New Orleans offers a unique opportunity to study restoration ecology. While any restoration project may take decades or even centuries to 'complete', the quality and efficacy of early management will influence the overall success of the project. Indeed, the success(ion), if you will, of this newly established coastal prairie, will require the continuous commitment from all stakeholders, as well as further research addressing all ecological, social, and design inquiries.

TABLE 4. List of Species by botanical name and their corresponding common names

	Species	Common names
FORB	*Alternanthera philoxeroides*	alligator weed
	Calyptocarpus vialis	horse mint
	Commelina communis	asiatic dayflower
	Coreopsis lanceolata	lanceleaf tickseed
	Dichondra carolinensis	dichondra
	Diodia virginiana	Virginia buttonweed
	Eupatorium capillifolium	dog fennel
	Eupatorium serotinum	lateflowering thoroughwort
	Helianthus mollis	ashy sunflower
	Oxalis spp	oxalis
	Phyllanthus urinaria	chamber bitter
	Pycnanthemum muticum	mountain mint
	Rudbeckia maxima	great coneflower
	Ruellia noctiflora	nightflowering wild petunia
	Rumex crispus	curly dock
	Salvia lyrata	lyreleaf sage
	Sida rhombifolia	Cuban jute
	Solidago canadensis	goldenrod
	Symphyotrichum pilosum	aster
	Trifolium L.	clover
GRASS	*Alopecurus spp*	fox tail grass
	Carex spp	sedge 1
	Cynodon dactylon	bermuda grass
	Cyperus spp	sedge2
	Digitaria haller	crab grass
	Paspalum spp	paspalum
	Poaceae spp	grass
	Schizachyrium nees	little blue stem
	Scirpus spp	scirpis rush
	Sorghum halepense	johnson grass
TREE	*Celtis laevigata*	hackberry
	Fraxinus pennsylvanica	ash
	Quercus nigra	water oak
	Quercus virginiana	live oak
VINE	*Campsis radicans*	trumpet creeper
	Cocculus carolinus	Carolina coralbead
	Ipomoea	morning glory
	Rubus spp.	blackberry

<u>Figure 4</u>. Location of study site is within the red boundary, located in City Park, New Orleans, LA.

Figure 5. A 50 cm square frame divided into four 25 cm squares used to estimate cover percentage. As the study plots were 3 m^2, the entire 50 cm^2 frame and the smaller 25 cm squares represented 2.78% and 0.69% cover percentages, respectively

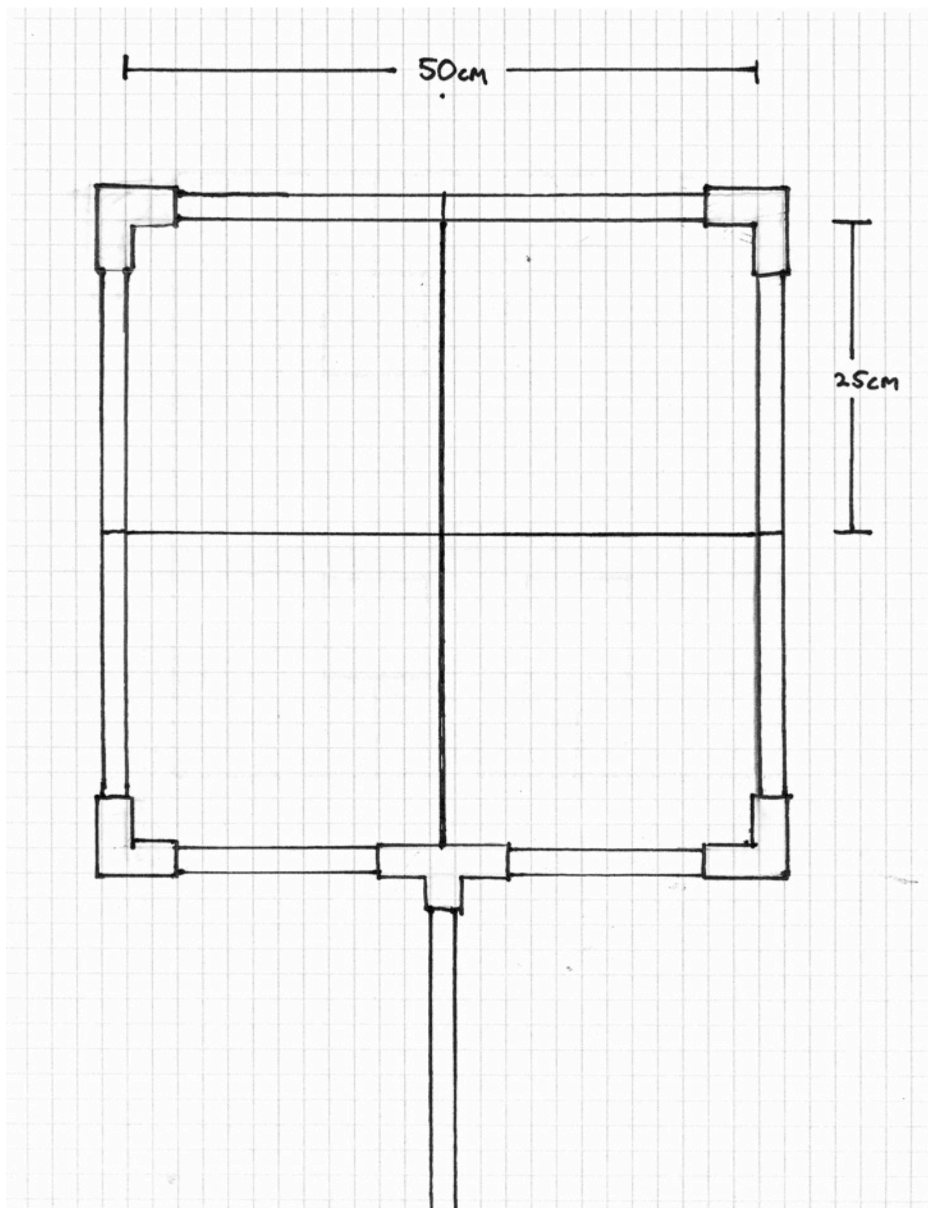
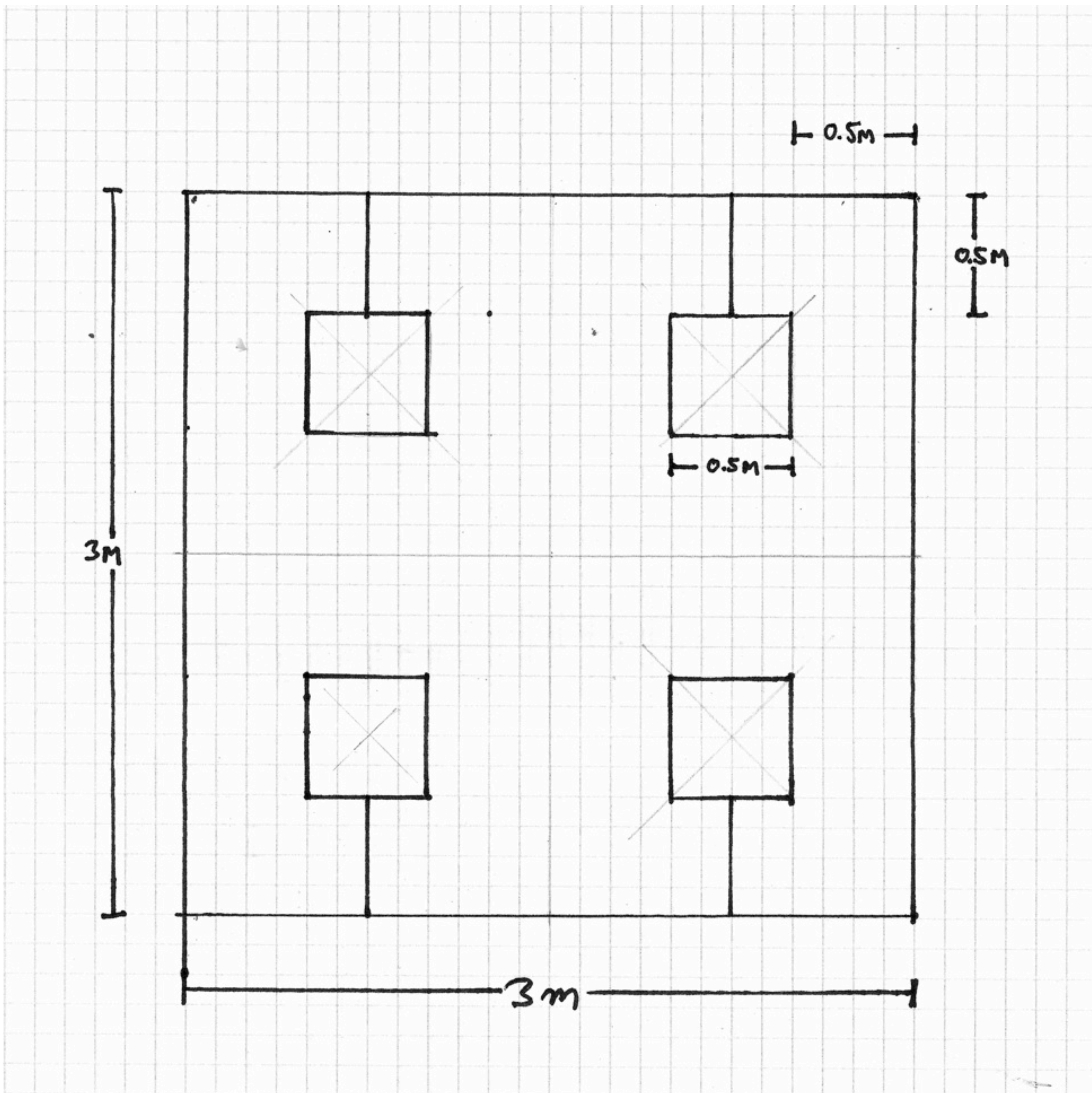


Figure 6. 3 m^2 Study plot with the four 50 cm^2 sampling plots. All study plots were sampled using this set-up.



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APPROVAL SHEET

This is to certify that Krisztian Megyeri has successfully completed

his Senior Honors Thesis, entitled:

The Impact of Ambrosia trifida (giant ragweed) on Native Prairie Species in an Early Prairie Restoration Project

Martin T. O'Connell — Director of Thesis

Mark A. Kulp — for the Department

Carl D. Malmgren — for the University Honors Program

December 6, 2011
Date