BIOCHAR AS A SOIL AMENDMENT FOR WILLOW AND COTTONWOOD PLANTINGS IN A RIPARIAN RESTORATION

By

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ABSTRACT

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Thesis directed by Professor Timothy R. Seastedt

Riparian areas provide some of the most valuable ecosystem services of any habitat type, both for native flora and wildlife, as well as human society. The global degradation of these critical systems makes their restoration imperative for the preservation of overall ecological integrity since depleted riparian corridors will have cascading effects that pervade upland and downstream regions alike. In western river systems there are two key limiting factors that can lead to failures during re-vegetation efforts along riparian stream banks: one is a lack of available soil nutrients, and the other is a lack of soil moisture. Biochar has been promoted as an effective soil amendment that increases both nutrient availability and soil moisture but to my knowledge has not been used in the context of riparian restoration. My experiment tested biochar as an effective addition to the slurry backfill of planted poles and container plants of two essential riparian tree species along Colorado's Front Range: *Populus deltoides* and *Salix exigua*. Convention dictates that poles planted for restoration purposes should be selected from dormant cuttings and planted in the early spring to ensure that environmental cues for growth do not yield premature budding until spring runoff has achieved an adequate supply of ground water to sustain long term survival. In practice however, early spring plantings can be logistically challenging and the ability to plant non-dormant cuttings later in the summer could make restoration efforts easier for land management agencies to complete. I used non-dormant poles to test whether biochar's effect on soil characteristics could offset the disadvantages associated with harvesting and planting cuttings in the late summer. My results showed no effects of biochar on the survivorship of either poles or container plants. Container stock of both *P. deltoides* and *S. exigua* survived better than poles of either species with or without a biochar treatment. Anecdotally, there was superior early growth in biochar treated plants, but this growth may have been phenologically maladaptive. Due to logistical constraints early in the planting process this growth could not be quantified.

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CHAPTER 1: OVERVIEW

The significance that riparian corridors play in the overall health of ecosystems far exceeds their magnitude in geographic size (Naiman and Decamps 1997). The plethora of services they provide to human society has made them susceptible to intense disruption and depletion at the expense of numerous species of flora and fauna worldwide (Naiman et. al 1993). In recent decades the restoration of riparian areas has emerged as a critical goal of land management agencies that are required to both ensure the sustainable access of the public to riparian environmental services, and to protect the wildlife and native vegetation that depend on them (Lake et. al 2007). Despite this imperative need for restoration, coordinated efforts to restore stream systems at the watershed scale remain challenging for scientists and engineers to organize and monitor (Lake et. al 2017), and many of the conventions pervading the restoration industry are not well supported by the available evidence (Palmer et. al 2010).

Initial efforts toward restoring riparian areas are largely centered around establishing native vegetation that represent appropriate successional stages at different levels of elevation from the water table (Giordanengo and Mandel 2016a). In western systems large flood events are an integral component of the historic hydrologic regime. The main limiting factors preventing native trees and plants from growing then, are not high magnitude disturbances, but rather the availability of soil moisture and nutrients (Goodwin et. al 1997). Biochar in the form of charcoal is a stable, carbon rich solid that has been used mainly for agricultural applications to enhance growth and fertility. Its propensity to enhance available moisture and nutrients in the soil, and to sequester toxins has made it useful for restoration purposes, particularly near former mine sites (Fellet et. al 2011). To my knowledge however, its effectiveness has yet to be determined in the context of riparian restoration. To these ends I conducted an experiment to find evidence as to whether a biochar soil application could potentially enhance the survivorship of two important trees species along the Front Range of Colorado, *Populus deltoides* and *Salix exigua*.

While convention for riparian plantings dictates the use of dormant cuttings planted in the early spring (Hoag, 2007), I used non-dormant cuttings and planted in the fall in order to find out whether a biochar soil amendment could potentially off set the disadvantages normally associated with this approach. Early spring plantings are often logistically challenging for restoration projects to complete, for example when water levels are too high to transport equipment to alternate sides of riverbanks, or when spawning and migration patterns of desirable wildlife may be disrupted by machinery or the presence of laborers. Large-scale restoration efforts undertaken in the late summer or fall may still take advantage of dormant poles, provided that the cuttings have been stored properly during the spring and summer, however this is often costly. An appealing option for land managers and contractors is to be able to plant non-dormant poles in the late summer to take advantage of less challenging conditions and to conserve financial resources.

The following chapters will provide background and context for my research project, as well as a summary of the experiment itself. Chapter 2 is a review of the literature surrounding biochar and its effects on soils, plant productivity, and environmental remediation. Chapter 3 is a summary of current trends in riparian and wetland restoration based on the peer-reviewed literature. Chapter 4 introduces the experiment I conducted, outlines the methods and results of the project, and concludes with a broader discussion of the context and ongoing progress of the restoration itself.

CHAPTER 2: THE EFFECTS OF BIOCHAR ON SOILS, PLANT PRODUCTIVITY, AND ENVIRONMENTAL REMEDIATION

Biochar has been defined as a product of organic thermal degradation in the absence of air and is distinct from charcoal in that it is used as a soil amendment (Lehmann et al. 2011). Some have suggested that it may be a key component to improving soil fertility, enhancing ecosystem services, and sequestering carbon from the atmosphere (Lehmann et al. 2006). This notion of long-term carbon storage was partly inspired by the discovery of so called "Terra Preta" or "dark earth" near the Amazon region in South America (Ahmad et al. 2014). The carbon pool in these soils is thought to persist for hundreds and even thousands of years (Sombroek et al. 2003) and the soils themselves were likely augmented or even engineered by ancient humans using slash and char techniques to sustain intensive agriculture in the poor, acidic, tropical soil (Barrow 2012). Biochar has since been investigated for a growing number of useful properties that show promise to improve agricultural yields, mitigate soil degradation, and produce energy as a byproduct of its manufacture. The effects of biochar on a given system are highly variable and dependent, not only on the ecology of the system to which it is applied, but also on the material used to produce the biochar (feedstock), as well as the method of production (pyrolysis temperature and rate). While studies suggest that biochar additions to soils could have a wide array of short-term benefits, the long-term effects of large-scale applications that would be necessary for effective global carbon sequestration remain unclear and potentially concerning from an environmental standpoint. The economic feasibility of such a scenario is also not entirely explicit. However site-specific uses of biochar appear to have few drawbacks and may be excellent ways to purify water sources and remove toxic pollutants from soils. Additional research and the careful systematic documentation of how biochar production affects select properties for precise uses will help increase the efficacy of this trend. Here I conduct a literature review featuring current research into the effects of biochar on soil and plant productivity, as well as its potential application toward environmental remediation.

PLANT GROWTH

A growing number of studies have demonstrated that the overall effect of biochar soil applications on plant growth is generally positive (Sohi et al. 2010). The complexity of interactions between soils, water, plants, and microorganisms make it hard to isolate the mechanisms by which biochar assists in this process. However, it is generally accepted that several synergistic effects of biochar treatments on soils facilitate plant growth indirectly. These include nutrient and water retention, improvements in soil pH, increased cation exchange capacity, the promotion of mycorrhizal fungi, the improvement of microbial populations and functions, and resistance to microbial pathogens (Elad et al. 2012 and Elad et al. 2010). In general biochar appears to have the greatest effect on plant growth in poor, acidic, tropical soils (Atkinson et al. 2010) and may even be effective in alleviating salt induced stress on plant growth in saline loaded environments (Thomas et al. 2013). A 2011 meta-analysis of 371 independent experiments showed that in spite of substantial variation in soil, climate, and char production methods, aboveground plant production and yield generally increased through the use of biochar soil treatments (Beiderman and Harpole 2013). At the same time, a large degree of variability was also reported for growth belowground, and in contrast to annuals, perennial plants did not have a significant response to the biochar treatments in general. Even among annuals plants, not all studies report positive growth effects after biochar treatments, with many encountering critical thresholds after which growth is retarded (Rajkovich et al. 2012). Several of the benefits incurred through biochar additions have been noted when fertilizer is added as an amendment in conjunction with biochar, leading some scientists to argue that biochar alone may not be sufficient as a soil amendment for initiating re-vegetation (Beesley et al. 2011). At the same time, there are ample studies demonstrating significant gains in productivity with isolated biochar additions (Shultz et al. 2015). The reason for this apparent discrepancy may be due to site-specific interactions between biochar, soil types, and climate (Jeffery et al. 2011; Haefele et al. 2011) as well as differences in the feedstock and pyrolysis temperature at which the biochar was processed (Sun et al. 2014).

While these gains in overall plant growth are encouraging for agronomic and restorative goals, the long-term effects of biochar as an additive are unclear. Some studies on agricultural crops have suggested that biochar soil additions have a dampened effect on plant growth over time. An experiment in Wales applied biochar to replicate field plots at three different rates with subsequent reapplications to certain plots over the course of three years. At the end of the trials the plots that had only received an initial biochar treatment did not see any improved crop performance or soil fertility as compared to the controls at the three-year mark (Quilliam et al. 2012). A second Wales study also reported only marginal gains in plant growth after a three-year trial with biochar treatments, and the gains appeared to be species dependent, with maize showing little to no response while grasses demonstrated small but significant increases (Jones et al. 2012). Experiments like these demonstrate the need for further study with respect to the long-term prospects of biochar as a solution to farming or re-vegetation efforts, and should be repeated in a wider variety of soils and climates. At the same time, few studies have reported any negative effects on soil quality, and the reported changes to system functioning are generally small and transitory suggesting that the potential gains of using biochar are not outweighed by the risks.

MICROBES AND SOIL BIOTA

Numerous studies have documented an increase in microbial population growth and the activation of dormant soil microorganisms following biochar additions to soils (Ameloot et al. 2013), however strong trends exist in the other direction making a firm conclusion regarding biochar's overall effect on microbes difficult to ascertain (Lehmann et al. 2011). When the increase of microbes does occur it may lead to increased soil respiration and an initial rise in CO2 flux (Smith et al. 2010). In terms of biochar's overall carbon footprint, this short-term release of CO2 caused by the breakdown of soluble organic carbon by microorganisms, as well as the release of carbonate contained in the biochar itself, is small in comparison to the long-term sequestration potential of mixing biochar into soils (Jones et al. 2011, Zimmerman et al. 2011). Some initial theories suggested that the inherent resistance to breakdown might prevent biochar from being a direct food source for microbes (Lehmann 2007b), but studies have since shown that direct microbial consumption is occurring to a certain extent (Kuzyakov 2009). The mineralization patterns show swift initial degradation of labile and volatile components, followed by the very slow degradation pattern that makes it a substantial carbon sink (Ameloot 2013).

On a structural level it has been suggested that biochars' porous texture, large internal surface area, and its adsorption of soluble organic matter provides ideal habitat for microorganisms to colonize, grow, and reproduce. Variation in pore size may provide micro-flora with protection from grazing, as well as increase the water-holding capacity of the soil (Thies and Rillig 2009). Many of the chemical changes to soil environments that are initiated through biochar additions are also conducive to improving the habitat of microorganisms. Observed increases in cation exchange capacity (Cheng 2006), increases in soil pH to buffer acidic soils, as well as increases in carbon content (Van Zwieten et al. 2010) all enhance soil habitat and potentially explain the observed positive relationships between biochar treatments and microbe population growth. Negative trends have been explained by some as the effect of biochar adsorbing heavy metals or organic compounds harmful to microbes (Lehmann et al. 2011). Properties of the biochar will be variable depending on the pyrolysis temperature at which it is produced, as well as the source material from which it is formed. Subsequently the effects of biochar treatments on microbial communities will vary with these conditions as well (Ameloot 2013). Biochar additions have also occasionally resulted in decreases of certain fungal microbes such as arbuscular mycorrhizal fungi (Warnock et al. 2010), although generally they tend to expand these populations (Warnock 2007). In terms of macro soil organisms earthworms have also had variable effects reported from biochar treatments. Some have suggested that earthworms play a key role in distributing biochar to different levels of the soil profile (Lehmann et al. 2011), while others have observed negative effects on earthworms resulting from the rise in pH that can occur with the addition of biochars produced from crop residues, sludges and manures (Ahmad 2014). The propensity for some microbes to benefit from biochar more than others under specific conditions, and for some not to benefit at all draws attention to the need for further study investigating the interactions between biochars of different feedstock and pyrolysis temperatures, as well as the ecological ramifications on the microbial communities that biochar additions are applied to (Lehmann et al. 2011).

WATER AND NUTRIENTS

Biochar’s high organic carbon content has prompted some scientists to propose it as a soil conditioner to improve physiochemical and biological soil properties (Ahmad et al. 2014). Water retention capacity, decreased nutrient leaching, and increased nutrient availability for plants are among the most important observed effects of biochar soil additions that contribute to overall soil health.

Studies have consistently demonstrated that biochar treatments allow for greater soil moisture content to collect and persist in target soils, an outcome which may increase in importance in the face of a drier, hotter climate (Atkinson et al. 2010). In light of this it has been suggested that biochar's ability to increase soil water retention capacity may be more effective in sandy soil than in soils with higher clay content (Woolf 2008). A recent study in a semi-arid region northeast of Brazil experimented with five biochars derived from agricultural waste to test if additions would increase water retention in the sandy, overused, moisture deficient earth. With the addition of just a 5% biochar mixture, water retention increased by up to 8% (Mangrich et al. 2015). A laboratory experiment in Japan tested water holding capacity in sandy soils, and found that biochar forged with a pyrolysis temperature of at least 500 degrees was able to increase saturated water content by 56% (Uzoma et al. 2011), while another study using a sandy loam from an Iowa State research facility reported a 23% increase in gravity drained soil water content as compared to controls (Basso et al. 2013). It should be mentioned that the capacity for biochar to retain water is not limited to sandy soils. Among many different other soil types, studies have demonstrated biochar induced water-holding capacity even in colder climates (Karhu et al. 2011) and temperate agricultural soils as well (Laird et al. 2010).

It has been suggested that this moisturizing effect on sandy soils may be a result of the increased total pore volume that biochar adds to the soil substrate (Abel et al. 2013). However the mechanisms behind biochar's tendency to increase soil moisture have not been decisively explained (Mangrich et al. 2015) and the effectiveness of any given treatment appears to be highly dependent on the feedstock and pyrolysis temperature (Singh et al. 2010). One study in the Netherlands did not report increased water holding capacity in a sandy soil following the addition of a slow-pyrolysis biochar produced at 400 and 600 degrees (Jeffery et al. 2015a). The authors commented that many of the studies suggesting that intrinsic structural and chemical properties of biochar enhance soil moisture content are using application rates that are not sustainable for large-scale field implementation. Other studies in soils with heavier clay content have also reported outcomes failing to demonstrate biochar-induced increases in water retention (Hardie et al. 2015) warranting further research into how various combinations of biochar feedstock and production will affect different soil types.

In addition to water retention, biochar's capacity for retaining available nutrients in soil is thought to be one of its essential properties that facilitate plant growth (Atkinson et al. 2010). Some have suggested that biochar may provide nutrients directly to plants in the soil, as well as contribute indirectly to plant growth through decreasing nutrient leaching (Chan and Xu 2009). Biochar is generally reported to increase soil pH (Rees et. al 2014) which, in addition to increasing available soil nitrogen and phosphorus, can immobilize heavy metals and potentially make it an effective treatment for contaminated soils (Houben et. al 2013). A collection of studies has demonstrated that biochar can act as a reservoir for phosphorus and that a measure of that phosphorus is available for uptake by plants (Yao et al. 2013). A recent study in southern Queensland Australia analyzed the ability of 9 different biochar types, both natural and manufactured, to retain phosphorus and improve phosphorus availability in soils. The results reported not only that biochar additions increased phosphorus availability through direct inputs, but also facilitated phosphorus availability indirectly through the retention of nutrients in applied fertilizer (Zhang et al. 2016). In contrast to biochar's influence on phosphorus availability, its effect on soil nitrogen cycling appears to be more variable. For example some short-term studies on nitrous oxide emissions report no reduction (Karhu et al. 2011) while others report significant decreases (Zhang et al. 2010). A 2013 review of biochar and soil nitrogen dynamics concluded that biochar generally appears to reduce inorganic nitrogen leaching, nitrous oxide emissions, and ammonia volatilization, while at the same time potentially increasing biological nitrogen fixation (Clough et al. 2013). This last point is somewhat unclear as many studies report the opposite effect, that is that biochar decreases available nitrogen in a system potentially through adsorption or increased microbial activity and subsequent nitrogen immobilization (Nelson et al. 2011). Other studies have supported the notion that biochar can enhance available soil nitrogen (Rondon et al. 2007) making further study necessary before strong conclusions can be drawn.

PRODUCTION AND ECONOMICS

Over the last decade the combination of biochar production and soil application has been touted as a strong measure to mitigate climate change (Lehmann et al. 2006). There are five main processes by which it is produced; slow, fast, and flash pyrolysis, torrefaction and gasification, each of which can yield a useful bi-product in the form of a solid (char), liquid (bio-oil) or gas (syngas) and each of which has a plethora of applications toward environmental remediation (Brewer 2012). To a large extent the process of biochar production will dictate the essential properties that make a given biochar appropriate for a specific applied scenario. Using different pyrolysis rates and temperatures, as well as different feedstocks can produce unique biochars with nuanced variance that may need to be appropriately measured and cataloged in order to facilitate large-scale societal impacts (Manya 2012). Production procedures may not always be standardized, and consensus on the most efficient output and utilization is not always clear. For example several studies have reported that increases in pyrolysis temperature will increase biochar carbon content in the soil and subsequently total carbon sequestration (Azargohar et al. 2014; Bruun et al. 2011; Bruun et al. 2012) while others have advocated for lower temperature, slow pyrolysis as a more energy efficient strategy for balancing energy production and carbon storage to reduce greenhouse gases (Gaunt and Lehmann 2008; Downie 2011). Some have argued that the most important factor for maximizing sequestered carbon is the degree of chemical stability in the biochar produced since less stable forms are likely to return carbon quickly to the atmosphere (Crombie et al. 2013). At the same time processes yielding more stable forms of biochar are likely to produce less total biochar (Sun et al. 2014). The resolution of these kinds of differences in perspective may be necessary before the long-term environmental and economic feasibility of global biochar energy production and soil treatments can be appropriately evaluated.

The most promising usage for biochar soil amendments has been cited as an amendment for dry, depleted, tropical soils (Atkinson et al. 2010) and many have suggested that there is both natural and financial promise for biochar soil additions in poorer regions of the globe (Duku and Hagan 2012). However some scientists have pointed out that various analyses of the economic potential for large-scale biochar use in energy production and carbon sequestration seems to have demonstrated only marginal viability that may be dependent on the assumption that it is appropriate for long-term agricultural usage. Field applications may be sensitive to market uncertainties that relate to the benefits of soil applications and the duration of those benefits (Spokas et al. 2012). Others have pointed out that large-scale application of biochar-related remediation measures may depend on carbon market benefits from fossil fuel displacement. Lacking financial incentives, power companies may not choose to save char for soil application and instead combust it for abundant added energy. Farmers may also be unwilling to endure the short-term cost and effort of integrating biochar onto their farms unless there is appropriate compensation (Brewer 2012). Some incentives may come from biochar made specialty materials such as photovoltaics and fuel cells (Nanda et al. 2016) but how these industries might factor into a comprehensive economic outlook is unclear.

Questions have also arisen regarding potential public health and environmental concerns that could emerge from the large-scale usage of biochar as a soil amendment, such as soil degradation, erosion, stripping of forest resources for feedstock, and the extensive production of mutagenic and carcinogenic pollutants during pyrolysis (Quilliam et al. 2012). Contaminants introduced to the soil, either through toxic feedstock or through byproducts of biochar production itself, could potentially persist for many years (Jeffery et. al 2015b). Any advantage gained from sequestering carbon would be lost if the mitigation is only replaced by other concurrent problems in land degradation and human health. Strong standardized regulations may be required in order to avoid this kind of practical dilemma.

ENVIRONMENTAL REMEDIATION

Substantive claims surrounding biochar and its role in environmental remediation have involved calculated predictions of substantial reduction in greenhouse gas emissions obtained from the dual function of providing energy alternatives to fossil fuels, and then storing carbon in soil as an amendment (Lehmann 2007a). Since the suggestion of this principal numerous studies have confirmed the long-term carbon sequestration potential of biochar despite its propensity for causing an initial short-term increase in $CO2$ flux (Zimmerman et al. 2011). However a recent meta-analysis of 91 published papers has challenged this notion, reporting that while biochar reduced soil $N2O$ emissions by over 30%, it increased soil $CO2$ flux by over 22% and had no effect on CH_4 flux. At the same time the degree of CO_2 flux was suppressed by the presence of fertilizer, indicating that agricultural applications may not contribute the same degree of carbon to the atmosphere, and all three greenhouse gas flux rates were significantly influenced by the biochar feedstock and pyrolysis temperature, as well as the texture of the soil it was applied to (He et al. 2017). These correlations draw attention to the high degree of variance in biochar, both in the physical and chemical properties that depend upon how it was produced, and the bio-chemical effect it has wherever it has been applied. Growing concerns about the ecological implications of wide spread global biochar treatments have recognized the need for greater understanding of specific individual biochar properties when considering biochar soil amendments for a given land management strategy, especially since it is likely to persist in whatever system it is introduced to (Ippolito et al. 2012). Other challenges to biochar as a reliable strategy for reducing global greenhouse gas emissions have revolved around questions about whether biochar energy production will be able to achieve a sustainable balance with biochar soil amendments within the current geo-political landscape (Woolf et al. 2010). Since standardized methods of production seem hard to come by and demand is likely to be much higher for energy production than for soil amendments, this proposed balance may be an unlikely prospect.

The majority of research into biochar related soil dynamics has focused on agricultural factors, but the imperative need for understanding biochar's long-term environmental impact is even greater in natural, uncultivated areas. A study in a Norwegian boreal forest reported a high degree of variability in decomposition rates of biochar both above and belowground, substantially influenced by feedstock and pyrolysis temperature (Kasin and Ohlson 2013), and a natural grassland study in the Netherlands reported significant alterations in plant community composition and nitrogen fixation patterns after a biochar soil addition (de Voorde et al. 2014). More study is desperately needed in the area of protected lands before biochar should be widely applied to an extent that will make a difference in global carbon levels. Variability in an agricultural environment may be substantial, but it is unlikely to match the variability in wilder settings, and any perturbations to a natural system will be much harder to address once the biochar has been added. If scientists and land managers want to use biochar to meet target goals in the environmental restoration of protected lands then predictable results are essential and appear to be severely lacking in the literature at this time.

Despite gaps in research, there remain ample studies documenting encouraging aspects of biochar treatments that can help mitigate environmental degradation of many kinds. Among the foremost reported is biochar's ability to absorb heavy metal and organic pollutants in soils (Tang et al. 2013). One study featured a dairy-derived biochar that was applied to soils laden with Pb and the noxious herbicide Atrazine. Biochar applications reportedly reduced the amount of each toxin by 100% and 77% respectively (Cao and Harris 2010). A collection of other studies confirm biochar's propensity for reducing the ability of plants to uptake herbicides in soil (Yu et al. 2009), the only drawback being that this trend simultaneously reduces the effectiveness of the herbicides, potentially necessitating increased overall herbicide use by farmers (Kookana et al. 2011). Biochar's tendency to absorb pollutants has also proved effective in mine reclamation. For example an application to a dumping site in Italy resulted in substantial reduction in mine tailings and toxic metals while increasing water holding capacity and green cover after planting restorative vegetation (Fellet et al. 2011). Ongoing studies at mines in the Upper Animas river basin in Colorado have also reported increased soil moisture content, decreased leachate concentrations, increased spatial vegetation cover, greater above ground biomass, and faster biomass accumulation following biochar treatments (Peltz 2011). Biochar may also be effective at treating discharged contaminants in groundwater including radioactive elements such as uranium (Xie et al. 2015), and despite some concerns about toxicity in the biochar itself, it may even sequester antibiotics from stream systems (Ippolito et al. 2012). While large scale use of biochar as a restorative element carries inherent risks, this should not deter continued research into its usefulness for a variety of environmental goals and strategies.

SUMMARY

Studies have demonstrated significant potential for biochar to increase plant productivity and reduce soil contaminants. Effects on soil biota are mixed and determined to a large extent by the feedstock and production methods by which biochar is derived. Several studies have clearly shown biochar's tendency to increase soil moisture content and cation exchange capacity, especially in poor, acidic, or sandy soils. Biochar also appears to assists directly with soil nutrient retention, securing phosphorus in an available form for plant uptake, and reducing greenhouse gas emissions by locking up nitrogen. Initially biochar additions may stimulate greater CO2 flux, but the greater trend is toward substantial carbon sequestration over the long term. The effects of biochar as an agricultural soil amendment have yielded mostly positive results with very few significant drawbacks on overall soil health. However variable effects of biochar on soil in natural areas makes further research into its large-scale application critical. Despite uncertainties regarding large-scale usage, biochar treatments are often reported as highly effective at reducing pesticide uptake in plants, purifying contaminated water sources, and reducing pollutants. The economic feasibility of wide spread biochar application to reduce greenhouse gas emissions is not promising but may be possible if appropriate economic incentives and regulations are implemented, including carbon market benefits and agricultural subsidies. In addition, public health concerns regarding toxicity within certain feedstocks, air pollution from production facilities, and the stripping of arable land for feedstock resources have raised valid concerns about biochar as a long-term strategy for climate change mitigation.

CHAPTER 3: RIPARIAN AND WETLAND RESTORATION: ECOSYSTEM SERVICES, BIOLOGICAL MECHANISMS, AND ADAPTIVE MANAGEMENT STRATEGIES

Freshwater resources are essential to human survival and are being depleted and destroyed at alarming rates (Naiman and Dugeon 2011). Conservation alone will not suffice to sustain the ecosystem services currently required by human societies, making restoration an essential component of addressing freshwater ecosystem health and sustainability (Palmer et al. 2009a). Wetland and riparian systems offer a disproportion amount of value to society compared to other resources and are therefore at the forefront of restoration agendas worldwide (Erwin 2009). Unique mechanisms that facilitate flood and drought tolerance allow wetland and riparian vegetation to thrive under the highly variable conditions of terrestrial watersheds (Blom et al. 1990). Despite these intricate adaptations, many species may be unable to cope with the exotic invaders and the depleted water supplies ensuing from global climate change (Seavy et al. 2009). The stochastic and complex nature of freshwater systems makes it very difficult to maintain broad-scale ecological structure and function at a watershed scale while exploiting valuable services for human consumption (Hillman and Brierley 2005). Traditional approaches to restoration for both riparian and wetland areas have disregarded system-level processes and targeted individual functions and services at the expense of overall ecosystem integrity (Beechie et al 2010). Current efforts to integrate broad-scale projects to restore wetland and riparian habitat have persisted despite substantial resistance from various stakeholders. However data indicating significant results toward wetland and riparian habitat improvement is lacking (Palmer et al. 2014a). One of the largest obstacles facing restoration ecologists is a lack of communication between engineers, contractors, scientists, and land managers (Bernhardt et al. 2007). Many well-funded projects with broad community support are failing to set quantifiable metrics that can be monitored and reproduced making the evaluation of outcomes difficult or impossible to assess scientifically (O'Donnell and Galat 2008). Frameworks for setting measurable goals and planning for long term monitoring are common, but have not yet taken root effectively in the industry (Pander and Geist 2013). Despite the challenges faced by wetland and riparian restoration, there is evidence that the work to date has had some important beneficial impacts including the reduction of fine sediment, nutrient, and pesticide inflows, as well as increasing target stream organisms (Feld et al. 2011). Here I review the literature on current approaches to freshwater restoration in order to evaluate published research trends, as well as assess the relative success and failure of major projects to date.

ECOSYSTEM SERVICES

The global degradation of freshwater systems continues at the peril of human beings that are indispensably linked to the associated life-sustaining properties of riparian and wetland habitats. Anthropogenic impacts on freshwater ecosystems are more and more recognized by the scientific community as responsible for the rapid deterioration of biodiversity, hydrologic integrity, and human health (Naiman and Dudgeon 2011). In order to address the pervasive loss, not only of freshwater resources, but of many other varieties of natural capital, some scientists have advocated for careful monetary calculations estimating the economic value of these "ecosystem services" in order to appeal to government and corporate entities that may have underestimated the costs of losing, and the benefits of preserving the underlying health of biological systems (Costanza et al. 1997).

While riparian flood plains cover only 1.4% of land surface area, some have estimated that they account for over 25% of all terrestrial ecosystem services (almost 4 trillion US dollars per year), with disturbance regulation, water supply and waste treatment accounting for the bulk of the total (Tockner and Stanford 2002). In addition to their intrinsic cultural and aesthetic value, healthy rivers and streams provide a vast array of provisional and supporting services including the delivery and storage of water for human consumption, the uptake of access nutrients and pollutants by riparian plants and microbes, the reduction of soil erosion via stabilized bankside vegetation, and water purification through the reduction of suspended sediment (Palmer et al. 2009b). The importance of wetlands is similar in magnitude, with wetland land cover totaling less than 3% of the global area, and potentially contributing as much as 40% of global annual renewable ecosystem services (Zedler and Kercher 2005). Wetlands are critical to the health and sustainability of entire watersheds, supplementing through different mechanisms many of the same benefits that riparian systems offer, and adding a collection of unique values as well. They form critical habitat for sundry species of micro and macro fauna, recharge groundwater supplies, stabilize shorelines, reduce eutrophication risks, store and filter sediment in water columns, and substantially mitigate flood damage (Keddy et al. 2009).

Despite growing scientific and political interest in ecosystem services as a model for conservation and restoration there remain difficulties and potential dangers in the monetization of natural processes (Martin-Lopez et al. 2014). Ideally financial calculations yield results suggesting that natural systems should be preserved, managed, and restored at large scales that maximize ecological integrity. This is especially effective with riparian and wetland systems that are interdependent at a watershed scale. Studies supporting broad conservation and restoration goals are not uncommon, for example a 2012 study on coastal wetlands reported flood control values were underestimated by billions of dollars (Costanza et al. 2008), and a 2010 assessment of ecosystem services in the Mississippi Alluvial Valley reported that gains from combined restoration and conservation initiatives under consideration by the U.S. Wetland Reserve Program would substantially exceed public expenditure despite the extensive magnitude of the objectives (Jenkins et al. 2010). On the other hand placing a dollar value on nature forces contiguous ecological systems to be subjected to fluctuating markets dictated by supply and demand. A study in Wenzhou China comparing ecosystem service values between constructed and natural wetlands found that despite natural wetlands valuing higher in every single metric except one, constructed wetlands were more profitable to the state because of how much money was saved through the superior waste treatment of strategically placed urban facilities (Chen et al. 2009). Other studies, despite acknowledging the superior ecological value of natural wetlands, have also concluded that constructed wetlands present far more economic benefits to society (Yang et al 2008). While wastewater treatment is a highly effective use of constructed wetlands and may help to preserve more delicate natural environments (Vymazal 2010), the notion of isolating functions and services, and extracting products from complex freshwater ecological systems in order to meet market demands is concerning due to a lack of adequate information, and the potential for destroying overall system function (Robertson 2000). For example in comparison to targeted efforts, restoration at a watershed scale may provide far more direct environmental and societal benefits, such as decreased greenhouse gas emissions and increased nutrient retention (Thiere et al. 2011), but if these benefits have not been appropriately financially evaluated, or are unable to be quantified fiscally they may be overlooked, overexploited, or potentially even squandered in a market driven framework.

This tendency toward the monetization of freshwater systems and natural capital in general is paving the way for a growing ideology advocating for the discrete separation of individual constituents of ecological systems in order to facilitate environmental engineering for human benefit. Scientists in Sweden advocated for targeting specific wetland functions at the expense of overall system function in order to protect economic resources after reporting that shallow wetlands with high shoreline complexity are more likely to benefit waterfowl and other wildlife, while deeper wetlands are more likely to assist with phosphorous retention (Hansson et al. 2005). A study in Columbus Ohio revealed that the edge of a river in a bottomland hardwood forest produced higher methane emissions than constructed river diversion marshes prompting the authors to recommend designing future wetlands to reduce greenhouse gases rather than to match regional wetland structure and function (Sha et al. 2011). Others have literally suggested that restoration ecologists "forget natural references and focus on human benefits" (Dufour and Piegay 2009). Many studies are now polling public opinion in order to gauge which services people are willing to pay for and how much they are willing to pay, potentially misleading citizens into believing false paradigms where natural systems can be itemized and selected like commodities without repercussions on other facets of the environment (Carlsson et al. 2003; Holmes et al 2004; Milon and Scrogin 2006; Ojeda et al. 2008). Those concerned about this trend toward more industrial, anthropocentric approaches in conservation and restoration point out that environmental measures targeting individual services are very likely to degrade other services and functions that may be equally or even more essential than the desired outcome (Palmer et al. 2014b; Eulis et al. 2008). Even when engineered freshwater systems targeting individual services do not have any measurable adverse effects on other components, they are often functionally inferior overall, lacking adequate hydrologic flexibility to accommodate ecological variability in the landscape (Cole and Brooks 2000).

ADAPTIVE PHYSIOLOGY

In order to survive in a highly stochastic environment freshwater plants have evolved an array of critical adaptations that allow them to withstand seasonal, climate-induced high water flow. Both wetland and riparian plants may endure periodic flooding in lowland areas when snow is melting in the early spring or after major rain events. These conditions will impose a variety of physiological challenges that must be overcome internally (Colmer and Voesenek 2009).

The anoxic environment of plants during partial or total submergence will make the production of ATP very difficult and can result in tissue and functional damage if physiological and metabolic processes cannot be recovered when the plant is re-aerated (Gibbs and Greenway 2003). While waterlogged, the lack of light availability and carbon dioxide will also deplete sugar reserves needed to sustain energy production and to avoid cell damage and death (Bailey-Serres and Voesenek 2008). In addition, the conductivity of water in the roots may decrease (Holbrook and Zwieniecki 2003) while toxic elements in the soil may collect and build up in the root tissues (Foy et al. 1978). Adaptive patterns to address these hardships have been categorized into 2 broad strategies namely Low Oxygen Quiescence Syndrome (LOQS) and Low Oxygen Escape Syndrome (LOES) (Bailey-Serres and Voesenek 2008). Plants using a LOQS strategy are able to use ATP efficiently, produce enzymes that can make ATP without the presence of molecular oxygen, and to produce chemical components such as antioxidants, reduced phenolics, and enzymes that work against the effects of cellular breakdown once plant tissues are re-exposed to the air (Colmer and Voesenek 2009; Armstrong et al. 1994). Plants oriented toward a LOES strategy are able to change growth rate and direction of shoot organs that enable plant tissues to surface above floodwaters, maintain and generate internal anatomical structures such as aerenchyma that facilitate efficient gas diffusion and pressurized exchange, and can also maintain and generate exterior features, such as leaves with thinner epidermal cell walls, that promote gas exchange between plant tissues and the waterlogged environment (Colmer and Voesenek 2009; Bailey-Serres and Voesenek 2008).

Several trees species are also well adapted to periodic flooding, some of which are major components of wetland and riparian restoration efforts (Stromberg 1993). The genera *Salix* and *Populus* within the family Salicaceae in particular comprise an extensive number of invaluable species used for environmental restoration as well as commercial utilization. Salicaceae vegetation dominates the northern temperate zone riparian floodplains (Malanson 1993). Many species are globally in demand for targeted environmental usage and currently cover over 80 million hectares worldwide (Ball et. al 2005). Their rapid growth rates make them ideal for agroforestry plantations that produce pulp, paper, and cardboard products in addition to sequestering carbon. Their ability to tolerate and counteract the effects of heavy metals in soils has also made them strong candidates for phytoremediation projects (Tognetti et. al 2013). In the context of riparian restoration, extensive root systems aggregate soil particles to establish solid stream banks that prevent erosion through shear stress reduction, and help to assemble appropriate levels of valuable free-flowing soil sediments from the water column. Their rapidly establishing vegetation also provides habitat for wildlife including ground nesting birds and small mammals, and at the same time it offers a food source for foraging ungulates such as deer and elk. Many species such as cottonwoods (*Populus*) and peach-leafed willows (*Salix*) grow large enough to form canopy structures that overshadow stream banks. The shade provided ensures that water and stream bank temperatures are appropriate for sundry macro-invertebrates, subsequently supporting valuable species of fish that feed on them. It also regulates temperature in the hyporeic zone where other micro-invertebrates thrive. Falling leaves and other debris from the canopy then act as organic inputs that feed these stream organisms and contribute to overall ecological function (Giordanengo and Mandel 2016b).

In western systems cottonwood and willow trees have adapted to periodic flooding in early spring followed by subsequent droughts in late summer and early fall (Amlin and Rood 2002). Physiological responses by cottonwoods to drought include reduced root and shoot growth, reduced leaf area and increased senescence, reduced trunk expansion and branch sacrifice, and crown die back (Rood et al. 2003). Despite these adaptations isotopic analysis has indicated that cottonwoods are heavily dependent on a consistent supply of alluvial groundwater originating from river flow (Busch et al. 1992) making them extremely susceptible to damming and other water diversions that can reduce the fitness of many phreatophytic species (Rood and Mahoney 1990). Willows are early colonizers in western riparian regions due to their ability to emerge in nutrient-limited oligotrophic areas, their prolific seed production, and their associations with critical mycorrhizal fungi. Key traits that have made willows ideal for restoration efforts include efficient nutrient uptake, tolerance of flooded and saturated soils, superior growth and re-establishment from severed cuttings, and the ability to absorb high levels of heavy metals (Kuzovkina and Quigley 2005). Willow seeds are viable for a limited portion of the year and require ample moisture to germinate during that time (McLeod and McPherson

1973). Like cottonwoods, this makes them vulnerable to shifts in flood intervals or changes in the water table that may occur due to anthropogenic disruption or climate change.

Populus and *Salix* species often co-exist in the same habitat forming close ecological associations (Reichenbacher 1984). This is counterintuitive according to community ecological theory that predicts biotic filtering between species with recent shared ancestry (Savage and Cavender-Bares 2012). Both *Populus* and *Salix* have demonstrated superior growth rates when growing in close proximity to sympatric neighbors (Grady et. al 2017) illustrating the importance of the community related interactions between the two genera and the early successional species they support. *Populus* and *Salix* share a variety of traits related to their catkins that facilitate their important ecological role in riparian habitat establishment. Among these traits are preformation, which involves the development of structures before they are functional. *Populus* and *Salix* inflorescences emerge as soon as resting buds form, which may occur as early as May, and enables their catkins to establish before the reproductive buds break in the early spring. The main advantage of preformation is that it can facilitate precocity. Precocity, in the case of *Salix* and *Populus,* enables the maturation of reproductive buds before the development of vegetative buds, and ensures that flowering can occur before a large canopy develops overhead. For wind pollinated plants precocity offers the unique advantage of allowing greater air movement in between branches and vegetative tissues of nearby trees, increasing the chances of pollen reaching sexually mature flowers. For insect pollinated plants precocity offers a distinct competitive advantage for attracting bees during a time of year when other sources of nectar are not yet available. Characteristic floral reduction increases the efficiency of reproductive contact during these bee visitations by amassing numerous tiny flowers on a small landing surface without having to invest large amounts of energy into elaborate floral tissues. The result is often that both *Populus* and *Salix* species generate massive amounts of seeds each year. Bud dimorphism facilitates this precocity by effectively separating vegetative and reproductive meristems so that vegetative stems have no reproductive function and catkins have no vegetative function. An adaptive consequence of this is that *Populus* and *Salix* inflorescences are laterally distributed such that the emergence of an inflorescence will not arrest growth, enabling them to grow rapidly and establish early in comparison to competing species (Cronk et. al 2015).

The adaptive morphology of the catkin structure plays an important role in contributing to the ecological functions of *Populus* and *Salix* species. The enormous seed output coupled with early seasonal phenology will overwhelm riparian systems with a propagule pressure that is hard for other species to compete with (Braatne et. al 1996). For many species germination will occur very quickly, in some cases within 24 hours (Douglass 1995). Poplar and willow seeds can also germinate under a wide variety of temperatures and the seeds of some species manage to sprout in darkness or even when submerged in water without seedling deterioration (Karrenberg et. al 2002). The rapid vegetative growth facilitated by bud dimorphism extends not only to photosynthetic tissues that can outgrow competitors for light, but also descends to the root structures that aggressively expand downward to remain submerged in the declining water table following early spring runoff. Roots not only extend downward but also horizontally allowing plants to absorb resources from a large surface area and helping them to resist the erosive pressure of fast flowing water (Karrenberg et. al 2002). These root systems are able to form symbiotic relationships with a variety of arbuscular mycorrhizal fungi that benefit riparian soils including many neighboring plants species (Beauchamp et. al 2006). Many species can also regenerate from root and stem tissues, making asexual reproduction possible through the dispersal of severed fragments of any size. Large floods may then serve as a dispersal mechanism when branches, roots, or even entire trees are split apart and carried downstream (Karrenberg et. al 2002).

Altered hydrologic regimes that prevent periodic flooding and reduce ground water supply can lead to a number of physiological difficulties for many riparian species including growth suppression and mortality due to shifts in respiratory metabolism, reduced photosynthesis, and changes in mineral nutrition (Kozlowski 2002). Altered flow regimes in western systems have led to a rise in invasive species such *Tamarix ramosissima* and a decline in traditional gallery forest trees that are less capable of tolerating water stress and groundwater drawdown, and are not as fast to regrow after fires (Busch and Smith 1995; Smith et al. 1998). Several studies have reported the superior competitive ability of *Tamarix* species over native *Populus* and *Salix* for other reasons including increased leaf gas exchange and superior growth when water is abundant (Horton et al. 2001), a greater ability to withstand saline soil environments (Glen et al. 1998), and a greater ability to withstand rapid shifts in groundwater depth (Shafroth et al. 2000). Solutions to what appear as superior fitness traits are difficult to imagine. However a study in Fort Collins, Colorado found that fall flooding enabled *Populus deltoids* to outcompete *Tamarix ramosissima* the following year, suggesting that restoring flood regimes closer to historic levels may help bring back depleted native trees (Gladwin and Roelle 1998). Climate change will undoubtedly continue to challenge land managers and stakeholders to find ways of retaining native freshwater flora that are uniquely suited to survive in a given range of conditions that are rapidly disappearing (Capon et. al 2013).

STANDARDS, TECHNIQUES, AND RESTORATION RESULTS

Traditional approaches to river restoration have centered around channel reconfiguration and artificial bank stabilization to reduce erosion and runoff (Palmer et al. 2014a). This perspective is based on the theory that once streams can appropriately handle flow dynamics and sediment fluxes then other ecological processes (species assemblages, nutrient processing, decomposition, and primary production) will fall into place accordingly (Palmer et al. 1997). This assumption has been challenged aggressively by scientists wary of engineered landscapes that have targeted small channel reaches to protect infrastructure through a "command and control" (Holling and Meffe 1996) strategy rather than addressing stream restoration at the watershed scale (Buijse et al. 2002; Clark et al. 2003; Wohl et al. 2005). Frameworks for approaching holistic, watershed scale restoration based on ecological integrity are common (Kondolf 1995; Opperman 2010; Palmer et al. 2005; Pander and Geist 2013; Woolsey et al. 2007) but it has been pointed out that efforts to these ends are often difficult to implement due to conflicting priorities of stakeholders, lack of funding, or insufficient community support (Hillman and Brierley 2005). Despite these difficulties advocates have persisted in their emphasis of process-based restoration grounded in addressing the root causes of degradation, keeping projects relevant to the biological and physical characteristics of target sites, and avoiding restructuring natural areas in ways that exceed a given site's ability to sustain. Techniques used to accomplish this are wide ranging and include resurfacing or removing forest roads to reduce erosion and delivery of fine sediments, restoring hydrologic connectivity to accommodate historic high and low flow levels, replanting riparian forests to restore shade and nutrient inputs, reintroducing beaver to raise water table levels or increase sediment retention, and removing dams and levees to restore fish migration (Beechie et al. 2010). In contrast to this are the more widely applied techniques of using artificial riprap barriers to hold banks permanently in place, installing rock weirs that restrict the migration of aquatic biota, placing gravel beds for spawning in areas where none would naturally exist, and planting non-native riparian species in areas needing vegetation (Roni et al. 2008).

The watershed approach to restoration is rooted in the notion that projects should measurably improve not only the appropriate immediate indicators of ecological integrity, but also the stream's resilience, allowing for variable conditions and outcomes to unfold without disrupting key ecosystem elements and dynamics (Palmer et al. 2005). Wetland restoration ecologists have made similar arguments for some time, promoting a landscape perspective that factors in climatic variation and prioritizes the need for diverse wetland areas across a region in order to ensure that structural and functional components of a watershed have not been compromised (Bedford 1999). Wetland restoration has been particularly vulnerable to "command and control" approaches due to poor metrics applied in determining healthy wetland dynamics and characteristics (Maron et al. 2012) as well as a lack of understanding of their inherent role in providing ecosystem services (Maltby and Acreman 2011). In the United States these shortcomings have been exacerbated by the "no net loss" policy that allows for essential wetland systems to be destroyed and replaced by constructed wetlands that do not retain essential wetland traits (Zedler 1996). This is particularly alarming to scientists who believe that without natural reference wetlands available for comparison, restoration is largely incomplete and will emphasize only target functions that do not mitigate other losses in function at the watershed scale (Brinson and Rheinhardt 1996). To avoid this mistake landscape oriented wetland restoration ecologists have tried to initiate projects that allow for natural trajectories of succession rather than trying to speed up recovery for the purpose of meeting inadequate benchmarks, understand and facilitate the inherent pace at which different wetland ecosystem attributes develop, and address specific hydrologic regimes of individual wetland projects (Zedler 2000).

Regardless of how freshwater restoration is approached, it remains very difficult to assess outcomes for a number of reasons. One of the most pressing issues is the lack of established monitoring protocols within the industry. A 2007 study conducted 317 confidential interviews with project managers from riparian restoration projects across the United States and found that despite the fact that two-thirds of the managers felt the project they were working on was "completely successful" less than half the projects had measurable objectives, with post-project appearance and positive public opinion stated as the most common measure of success (Bernhardt et al. 2007). A similar study interviewed project managers working within the Upper Mississippi River Basin and found that out of 70 projects surveyed only 34% incorporated a quantifiable metric for success, using geophysical attributes and biological communities as criteria instead, with the latter rendered useless for future assessment due to a lack of pre and post project data collection (O'Donnell and Galat 2008). There have been numerous pleas by freshwater restoration ecologists calling for the incorporation of more overall scientific rigor (Palmer 2009a), an emphasis on hypothesis testing during project planning (Kentula 2000), and a uniform standard for monitoring protocols (Downs and Kondolf 2002). It's been difficult for these suggestions to synthesize into cohesive systems of collective cooperation (Bernhardt et al. 2005). Compounding the problem is the fact that even the projects that are prioritizing measurable objectives are not guided by industry wide standards, and more importantly are not sharing the information with other practitioners and stakeholders (Palmer et al 2007).

In addition to assessment-related challenges, there are also concerns about quantifiable reported outcomes. Surveys indicate little improvement in biodiversity and water quality for the majority of river restoration projects nationwide (Palmer et al. 2014a). This may be due to the emphasis on reach-scale endeavors (Bernhardt and Palmer 2011), or the use of techniques poorly informed by science, for example trying to mitigate substratum degradation without addressing the hyporheic zone (Mueller et al. 2014; Hester and Gooseff 2010). While successes at the watershed scale are not absent (Richardson et al. 2011), consistent results are rare even within high profile systems receiving large amounts of funding and publicity (Shields 2009), prompting some scientists to call for renewed efforts in conservation that prevent initial land damage in the first place (Roni et al. 2002).

There remain difficult and important questions surrounding the extent to which ecosystem function can be recovered at all after major disturbances and substantial human exploitation. Due to their complex, highly variable conditions freshwater systems are particularly vulnerable to disruption, and may easily be altered from their historic range of variability (Naiman et al. 1998). After major biotic or abiotic system changes have occurred, efforts to return to historic baseline conditions are difficult or even impossible in many ecotypes (Seastedt et al. 2008), and comparisons between restored and natural freshwater environments indicate that previous functions are either diminished or absent even after restoration (Zedler 2005). A comprehensive Massachusetts inventory of constructed wetland projects intended to offset the loss of natural wetlands found that almost 55% were not in compliance with state regulations for a number of reasons including insufficient size and hydrology, insufficient wetland plant cover, inappropriate species composition, and negative impacts on natural forested wetlands (Brown and Veneman 2001). A survey in Pennsylvania found that even constructed wetland projects that were meeting jurisdictional requirements were not retaining appropriate hydrologic structure and function, keeping sites inundated for too long and diminishing plant diversity (Cole and Brooks 2000). In Ohio researchers found that many constructed wetlands were in compliance with regulations only because the benchmark was restricted to the replacement of area covered by the wetland and did not specify additional metrics that contribute to many ecosystem services (Wilson and Mitsch 1996). While an argument can be made that monitoring periods may be too short to determine whether structure and function have been recovered in restored freshwater environments (Campbell et al. 2003) it appears that within currently accepted time intervals the data indicates that the "no net loss" policy adopted for wetlands by the U.S. government in 1989 is not working. Whether this is due to poor assessment methods, a lack of funding, insufficient system level knowledge, or combinations thereof, is unclear. What is clear is that restored wetlands do not match the hydrogeomorphic integrity of natural systems, making wetland conservation even more salient than previously considered (Whigham 1999).

Like wetlands, restored riparian systems also present particularly complex challenges when assessing whether ecosystem health and integrity have been reclaimed. The extreme heterogeneity of riparian dynamics is not scalable, stream ecology is governed by many different independent variables, and response times to treatments can be very long (Shields et al. 2003). This makes experimentation and assessment challenging, not to mention expensive, rendering results difficult to prove (Friberg et al. 2011). Riparian restoration has generally been considered unsuccessful for some time (Patten 1998) and despite some progress toward reducing fine sediment entry, reducing nutrient and pesticide inflows, and increasing target stream organisms, it remains difficult to find data that demonstrates long-term ecological recovery of function and services due to habitat enhancement (Feld et al. 2011). Perhaps the most effective strategy for restoring ecosystem services to freshwater systems is simply to reduce anthropogenic impacts. Since the early 1970's riparian forests have declined by over 35% in the United States (Jones et al. 2010). Riparian deforestation causes channel narrowing, reduces habitat, and undermines in-stream processing of pollutants (Sweeney et al. 2004). A river project in Denmark reported significant gains in biodiversity and other metrics by simply ceasing channel impoundments and restoring connectivity to a previously meandering river (Pedersen et al. 2007). A study on the Iberian Peninsula found that by limiting logging activity near headwaters and allowing dead wood to flow down river, the ensuing municipal savings from water purification and erosion control potentially exceeded profits from the logging operation itself (Acuna et al. 2013). In short, while restoration remains an essential component of addressing environmental degradation, it will never be a substitute for protecting in-tact systems, and is unlikely to deliver ecological benefits to the same degree as currently protected areas for many years to come.

SUMMARY

The field of wetland and riparian restoration remains critical to addressing the rapid degradation of the world's freshwater resources. While results toward these ends are slow and incomplete, the dire need to replenish damaged watersheds has initiated a massive push from scientists and land managers to integrate a comprehensive framework that will facilitate greater collaboration between researchers, stronger emphasis on science, and quantifiable outcomes that make industrial and municipal water usage sustainable across the globe. Because of the pace at which the global climate is changing, many riparian species are under threat of substantial decline or extinction. One of the most promising ways to address this concern is through efforts to return watershed systems to flow patterns that approximate historic hydrologic connectivity and flood regimes. Altered and restored freshwater systems do not provide equivalent ecosystem services compared to the degraded natural systems they are intended to replace. In this sense restoration should not be considered a viable alternative to conservation, but should work in tandem with efforts to protect watersheds to the greatest extent possible. In order to maximize the effectiveness of freshwater restoration there needs to be much greater communication between contractors, scientists, and stakeholders regarding project outcomes. To these ends, monitoring must take place far beyond current short-term benchmark standards. Project planning should include measurable goals with quantifiable and reproducible results. Future efforts toward freshwater restoration must incorporate hypothesis testing, precise experimental design, and rigorous long-term evaluation. Failure to do this may render the massive current influx of time, money, and resources devoted to rehabilitating riparian and wetland ecosystems effectively negligible on a global scale.

CHAPTER 4: BIOCHAR ADDITIONS TO SOILS OF COTTONWOOD AND WILLOW PLANTINGS FOR A RIPARIAN RESTORATION

INTRODUCTION

Riparian corridors play a disproportionately large role in the health and stability of a wide array of species and environmental processes (Naiman and Decamps 1997). Scientists have argued for some time that effective management and restoration of these areas could help mitigate many of the pervasive global challenges arising from land use and the subsequent degradation of environmental integrity (Naiman et. al 1993).

Historically western river systems have had to adapt to a variety of disturbance regimes that included high magnitude flood events resulting in the deposition of large sediment loads and geomorphic channel restructuring (Wohl, 2011). Current trends of exotic species to invade highly disturbed areas can potentially render these large flood events less functional on an ecological level than they were historically, and may add a substantial degree of risk in terms of invasive colonization (Diez et. al 2012). Anthropogenically altered fire regimes that contribute to frequent high-impact crown fires can lead to erosion and a decrease in buffering vegetation that subsequently augments stream power during large flood events (Benavides-Solorio and MacDonald, 2001). The effect of large fires on soils combined with the reduced overall annual stream flow due to human-altered hydrologic cycles may yield highly destructive changes in stream channel morphology during floods, compounded by a reduction in water table levels. This dynamic can create conditions in which native colonizers have difficulty competing with exotic invaders (Floyd et. al 2006). The flood of 2013 along the Front Range of the Rocky Mountains in Colorado fit this profile and may have provided competitive advantages to invasive colonizers over native species especially in areas experiencing a shortage of water and nutrients. While many native trees and plants of the Front Range are uniquely suited to thrive in riparian corridors after high magnitude flood events (Baker and Walford 1995), there are a wide array of exotics that are effective at outcompeting early colonizing native trees by acquiring and exploiting water at lower drawdown levels of the water table (Glenn and Nagler 2005), and disrupting key symbiotic relationships with abruscular and ectomycorrhizal fungi (Meinhardt and Gehring 2005).

Soil amendments may be one way to reverse this trend and shift the balance of early successional growing conditions in favor of native plants and trees. Biochar has been touted by scientists for over a decade as a natural solution for enhancing soil properties in order to facilitate both agricultural and restoration goals (Lehman 2007). As a soil addition it has proven effective at increasing soil microbes including beneficial fungi (Warnock et al. 2010) as well as increasing soil water holding capacity and nutrient retention (Karhu et. al 2011). To my knowledge, biochar's tendency to increase available soil nutrients and moisture has never been used in an applied restorative capacity on dry riparian soils, but may be able to neutralize the competitive advantage of exotics such as *Tamarix ramosissima* that frequently eclipse native cottonwoods and willows on desiccant, disrupted river banks.

In western systems the single most important element contributing to the success of riparian plantings during restoration efforts is water availability (Goodwin et. al 1997). Arid river corridors along the Front Range of Colorado are highly vulnerable to drought and reduced stream flow. This is especially true in areas where historic flood regimes have been altered for water management activities, and this can prevent the establishment of important native trees (Scott et. al 1997). Common practice for restorative riparian plantings is to harvest dormant cuttings from local riparian species and plant them in the early spring to take advantage of the spring melt and the plant's ability to withstand stress when emerging from winter (Hoag, 2007). However, this provides a narrow window of opportunity for agencies to select, harvest, and plant before the dormant period expires. High spring water crossings, as well as fish and waterfowl migrations may also present challenges to an early spring restoration project. Dormant cuttings have been documented as having superior survivorship. However non-dormant cuttings have been used effectively in riparian tree plantings, and this option may become more important in the face of a changing, warmer climate (Munda et. al 2005). Dormant cuttings remain a better option for restoration due to the physiological limitations of non-dormant cuttings with respect to their ability to emerge in spring ready to absorb and retain water and nutrients. However a biochar soil amendment was hypothesized to increase the ability of non-dormant cuttings to withstand the stress of the colder season and thrive as the temperature warms.

Container plants of various sizes are alternatives to cuttings such as whips and poles. Transplanted container plants are believed to be more likely to survive when compared to transplanted cuttings under many circumstances including when the risk of being dislodged from the ground is low. Cuttings remain a viable option however, since container plants are much more expensive and may present greater challenges during the planting process. They may even be preferable from a land management perspective because of the financial and logistic burden of purchasing, transporting and planting container plants (Hoag 2007). Parties integral to the completion of this research requested an investigation into whether container plants are actually more effective, even for small local restoration projects, when compared to cuttings from local trees.

My experiment endeavored to find evidence that adding a biochar soil amendment to the backfill of non-dormant pole cuttings can allow for successful late fall plantings of two important western tree species: Coyote Willows (*Salix exigua*) and Plains Cottonwood (*Populus deltoides*). I also tested the assumption that cuttings from local riparian trees are less effective in terms of survivorship than are container plants of the same species. To do this I planted 60 non-dormant cuttings and 60 container plants of both *S. exigua* and *P. deltoides* respectively in the late fall of 2017. One half of the plantings received a biochar soil amendment, while the other half was backfilled without it. I then measured survivorship in the early spring and again in the late summer to compare trees treated with a biochar soil amendment to controls, and to compare the survivorship of container plants to the survivorship of poles. My hypothesis was first, that biochar treated trees would show significantly greater survivorship than trees planted without a biochar soil amendment.   Secondly, I hypothesized that container plants would show greater survivorship than poles of the same species.

Municipal restrictions and logistical limitations prevented an additional treatment using dormant poles at the same study site. However, as an anecdotal comparison I planted 60 more trees (30 of *S. exigua* and 30 of *P. deltoides*) at a second study site in the early spring of 2018 and administered a biochar soil amendment to half the trees. My hypothesis was that trees which received a biochar soil amendment would show significantly great survivorship in the late fall, than controls.

The experiment was nested within a restoration project undertaken by the City of Longmont in Boulder County Colorado, in partnership with a grass roots restoration organization called Wildlands Restoration Volunteers (WRV). Restoration goals included jumpstarting a gallery forest to provide habitat for wildlife, and to stabilize the sandy soil substrate of the study site in order to protect nearby private property and downstream infrastructure from further erosion related damage and potential flooding.

METHODS

Study Site Descriptions

Study site 1 of the experiment is a section of a lower order reach within the St. Vrain Creek Watershed. It is located at the eastern edge of Boulder County Colorado approximately 1 mile south of the junction between East Ken Pratt Boulevard and East County Line Road. Sections of the creek at the site were blown out by the 2013 floodwaters and alternate channel routes were established in a meandering pattern. Substantial debris deposition occurred throughout the reach, although the relatively little infrastructure in the immediate area allowed for natural channel reconfiguration without significant damage. Large sand bar areas were formed and populated by early colonizing ruderal exotics. Cottonwood (*P. deltoides*) and willow (*S. exigua)* seedlings also aggregated near the toe and bank zones of the stream channels, as well as in microhabitats where the soil surface was fairly close to the water table. Planting areas for the experiment were selected for their potential to ensure survivorship and their potential to facilitate ecological benefits to the surrounding environment. A technical advisor affiliated with WRV was consulted regarding these planting locations. 2 major areas were partitioned for all the trees and were labeled the "Sand Bar" and the "Weedy Bench" respectively. Just before the project began in the fall of 2016 the surface of the Sand Bar was mostly bare ground and covered an area of approximately 2400 square meters. The water table was very close to the surface (within 1 meter) with a small, scattered collection of exotic weeds and first-year cottonwood seedlings emerging along the edges of the bar. The adjacent Weedy Bench area was approximately 30 meters north east of the Sand Bar covering approximately 200 square meters on an elevated berm that was almost 1.8 meters above the water table and overrun by an exotic species of the genus *Amaranthus*.

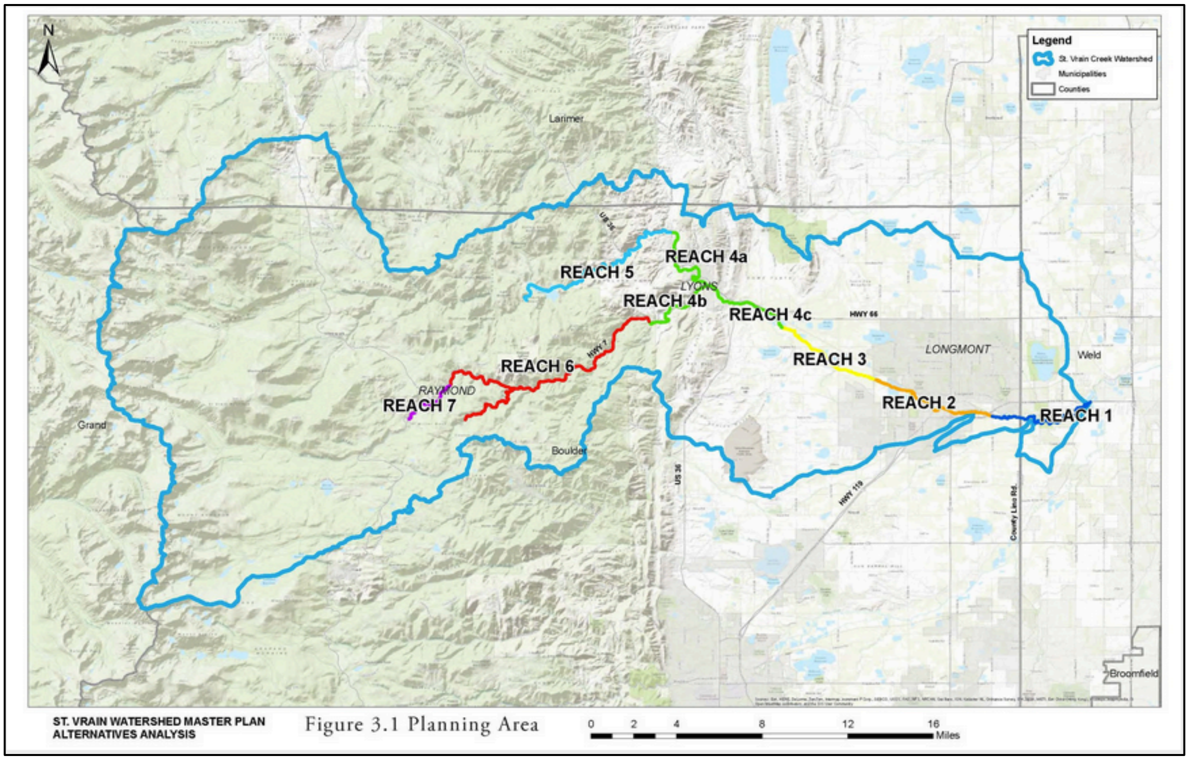

Figure 1 – Map of the St. Vrain Creek Watershed. Study Site 1 is located approximately in the middle of reach 1.

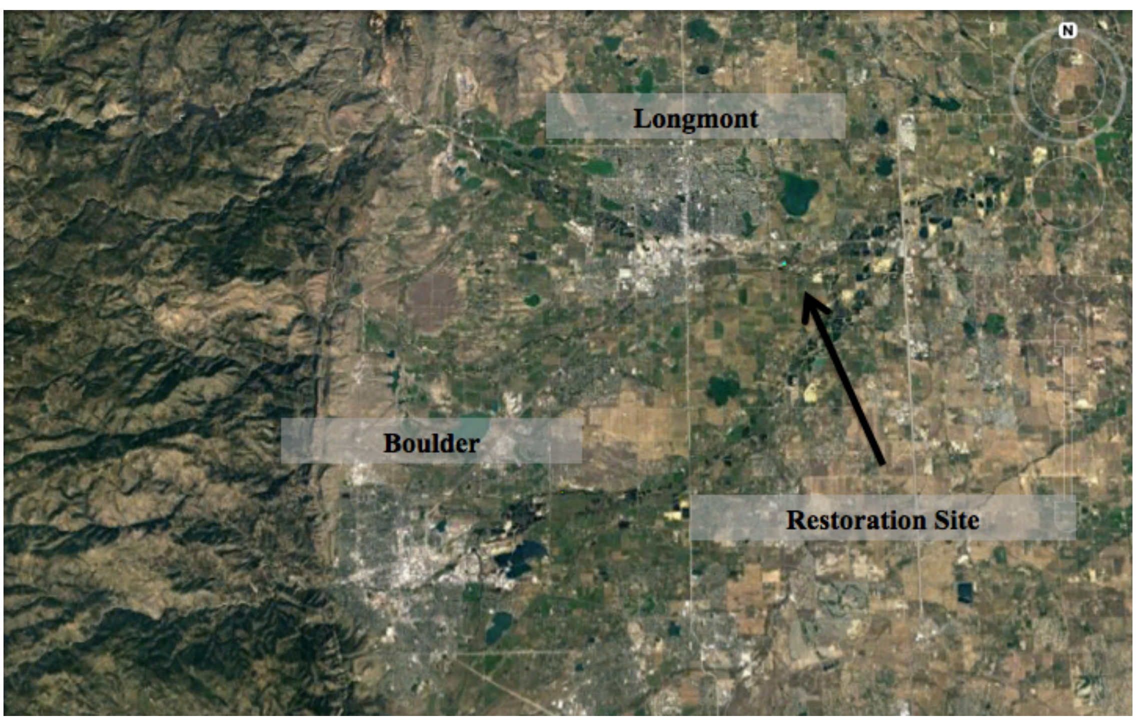

Figure 2 – Study Site 1 location in relationship to the cities of Boulder and Longmont Colorado.

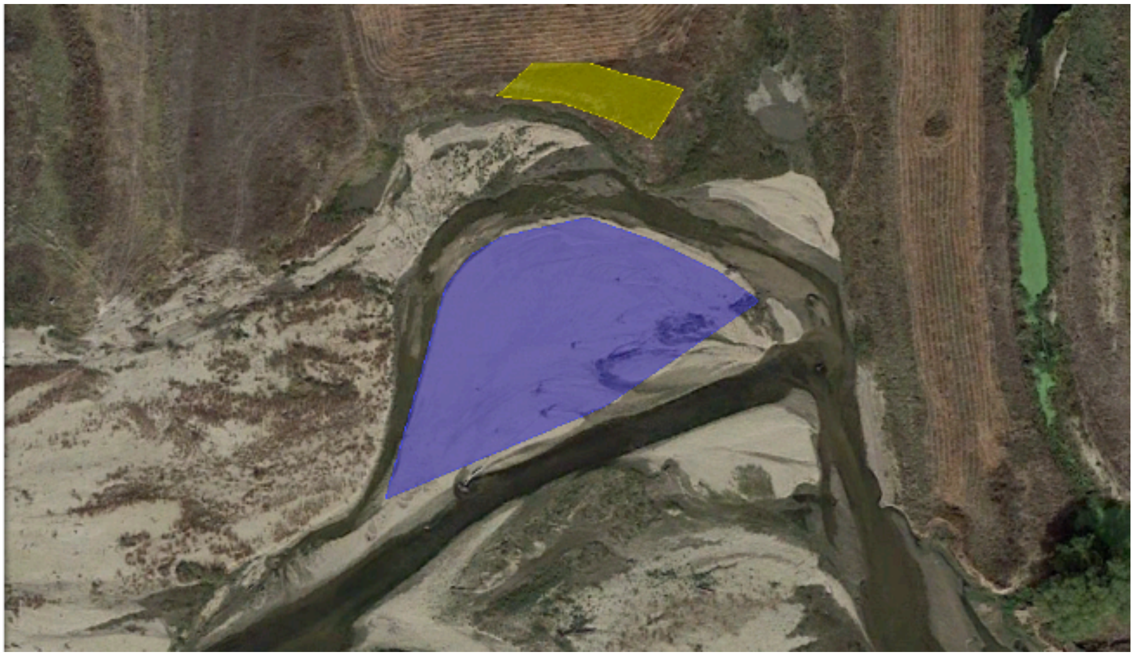

Figure 3 – Study Site 1 planting areas. The "Sand Bar" planting area is highlighted in purple. The "Weedy Bench" planting area is highlighted in yellow.

Study site 2 of the experiment is a small tributary of Lefthand Creek known as Spruce Gulch. The planting site is located on private property approximately 1.5 miles from the junction of route 36 and Lefthand Canyon Drive.

Figure 4 – Study Site 2 located on private property off of Lefthand Canyon Drive.

Figure 5 – Study Site 2 planting area in Spruce Gulch.

Materials used

For Study Site 1, 120 trees were planted in total along the Sand Bar and the Weedy bench combined. 60 of those trees were grown at the Aquatic and Wetland nursery in Fort Lupton Colorado, for a total of 30 cottonwoods (*Populus deltoides*) and 30 willows (*Salix exigua*) respectively. The 60 remaining trees were harvested 2 weeks prior to the project onset from local *P. deltoides* and *S. exigua* trees with the permission of the City of Longmont. The cuttings were secured using pruning shears and selected for vigor and appropriate size. *P. deltoides* cuttings were 4-5 meters tall and *S. exigua* cuttings were at least 3 meters. They were taken from multiple plants to ensure diversity, however the sex of each plant was not taken into account. The cuttings were prepared according to guidelines in Giordanengo and Mandel 2016b, and then soaked in an adjacent wetland for 2 weeks prior to the commencement of the project in mid-September.

Figure 6 – Soaked P. deltoides and S. exigua cuttings near the bank of a wetland area adjacent to Study Site 1.

For Study Site 2, 60 trees were planted in total. All 60 trees were dormant cuttings harvested with permission on property managed by the City of Longmont. Half the trees were *S. exigua* and half were *P. deltoides*. The dormant tree cuttings were harvested in the late winter of 2018, soaked within a walk-in water cooler set at 4 degrees Celsius and planted 2 weeks later.

Planting scheme and treatments

The Study Site 1 Sand Bar planting area contained 90 of the 120 total trees planted. 60 of those trees were *S. exigua* and 30 were *P. deltoides*. The 30 *P. deltoides* were all container stock grown by Aquatic and Wetland nursery. 30 of the *S. exigua* were also grown by Aquatic and Wetland nursery, and the remaining 30 *S. exigua* were cuttings harvested locally on city property. The arrangement of planted trees featured the larger growing *P. deltoides* occupying the center of the area and the shrubbier *S. exigua* placed along the periphery. Container stock and cuttings were distributed randomly throughout the planting area to control for spatial variables affecting survivorship. Spacing between the *S. exigua* was at least 2 meters, while spacing between the *P. deltoides* exceeded 3 meters. Work was undertaken in the late September of 2016. Survivorship was recorded in the early spring and late summer of 2017.

Figure 7 – Sand Bar planting area on project day.

The Study Site 1 Weedy Bench planting area contained the remaining 30 trees, all of which were locally harvested *P. deltoides* cuttings. Trees were planted roughly equidistant from one another, spaced approximately 3-5 meters apart in a random distribution.

Figure 8 – Auger with extension poised for digging in the Weedy Bench planting area.

For the dormant cuttings planted at Study Site 2, the *P. deltoides* and the *S. exigua* poles were planted in separate, equidistantly spaced groups respectively along the toe region of the Lefthand Creek tributary. All cuttings were planted 12 inches into the water table. Work was undertaken in the late winter of 2017. Survivorship was recorded in spring and the late summer of 2017.

Figure 9 – Biochar treated and control S. exigua cuttings planted at Study Site 2.

At Study Site 1 trees were planted using a 300-series Home Depot 2-person auger. Holes were 18 inches in diameter and penetrated into the water table for all trees planted along the Sand Bar. A 3-foot extension enabled holes to be dug as deep as 1.8 meters for the *P. deltoides* trees planted on the Weedy Bench. For many of the *P. deltoides* trees planted on the Weedy Bench, this depth was just barely enough to reach the water table. As specified in Giordanengo and Mandel 2016b, backfill soil was made into a slurry using buckets of water collected from the adjacent St. Vrain Creek. For half the trees a 10% mixture of biochar was added to the backfill soil slurry. The biochar was acquired from the Biochar Now facility in Berthoud Colorado. We used a fine-grain mixture from beetle-killed pine tree feedstock, produced at a pyrolysis temperature between 550 and 600 degrees Celsius for 10 hours. In total there were 8 categories of tree planting types (4 per tree species), including 4 treatment groups, (2 per tree species). The *P. deltoides* planting groups were: 1) *P. deltoides* cuttings with a biochar treatment, 2) *P. deltoides* cuttings without a biochar treatment, 3) *P. deltoides* container plants with a biochar treatment, and 4) *P. deltoides* container plants without a biochar treatment. The *S. exigua* planting groups were: 1) *S. exigua* cuttings with a biochar treatment, 2) *S. exigua* cuttings without a biochar treatment, 3) *S. exigua* container plants with a biochar treatment, and 4) *S. exigua* container plants without a biochar treatment. Planted trees were tagged with numbered metal tags fastened to each tree trunk with plastic zip ties and logged for their relative position along the planting scheme. Trees were then monitored periodically until the early spring when survivorship was recorded for each individual tree. A second survivorship tally was made in the late summer.

For Study Site 2 trees were planted with a sharp-shooter planting shovel directly into the soil lining the creek bank. A soil slurry was used to backfill each tree using exactly the same procedure as was done at Study Site 1. Half the trees received a 10% mixture of biochar as part of the backfill soil slurry. In total there were 4 planting types among the 60 trees planted: 1) *P. deltoides* cuttings with a biochar treatment, 2) *P. deltoides* cuttings without a biochar treatment, 3) *S. exigua* cuttings with a biochar treatment, and 4) *S. exigua* cuttings without a biochar treatment. Trees were then monitored throughout the spring and survivorship was recorded in the late summer.

Statistical methods

In order to determine whether the biochar soil amendment enhanced survivorship and whether container plants were more likely to survive than cuttings, a chi-squared goodness of fit analysis was used for 8 pairwise comparisons measuring survivorship at Study Site 1 for both the early spring and late summer: 1) *P. deltoides* cuttings with a biochar treatment vs. *P. deltoides* cuttings without a biochar treatment, 2) *P. deltoides* container plants with a biochar treatment vs. *P. deltoides* container plants without a biochar treatment, 3) *S. exigua* cuttings with a biochar treatment vs. *S. exigua* cuttings without a biochar treatment, 4) *S. exigua* container plants with a biochar treatment vs. *S. exigua* container plants without a biochar treatment, 5) P. deltoides container plants with a biochar treatment vs. P. deltoides cuttings without a biochar treatment, 6) P. deltoides container plants without a biochar treatment vs. P. deltoides cuttings without a biochar treatment, 7) S. exigua container plants with a biochar treatment vs. S. exigua cuttings with a biochar treatment, and finally 8) S. exigua container plants without a biochar treatment vs. S. exigua cuttings without a biochar treatment.

For Study Site 2 a chi-squared goodness-of-fit analysis was used to determine whether a biochar soil amendment increased survivorship for dormant pole cuttings using 2 pair wise comparisons: 1) *P. deltoides* cuttings with a biochar treatment vs. *P. deltoides* cuttings without a biochar treatment, and 2) *S. exigua* cuttings with a biochar treatment vs. *S. exigua* cuttings without a biochar treatment.

Survivorship was confirmed in both groups as the presence of vegetative growth sustained since the time of the initial planting.

RESULTS

Study Site 1 – Early Spring 2017 (Cottonwoods)

Early April survivorship for the *P. deltoides* container plants was 100% for both the biochar treated plantings and the controls, yielding no significant difference between the two groups. Biochar treated *P. deltoides* cuttings showed slightly higher survivorship (13 out of 15) than did the *P. deltoides* cuttings without a biochar treatment (11 out of 15), but was not statistically significant ($p = .68$). *P. deltoides* container plants with a biochar treatment showed greater survivorship (15 out of 15) than the *P. deltoides* cuttings with a biochar treatment (13 out of 15), but the results were not statistically significant ($p = .7$). *P. deltoides* container plants without a biochar treatment also showed greater survivorship (15 out of 15) than *P. deltoides* cuttings without a biochar treatment (11 out of 15) but was not statistically significant ($p = .43$).

Study Site 1 – Early Spring 2017 (Willows)

Early April survivorship for the *S. exigua* container plants was equally high for both the biochar treated plantings (14 out of 15) and the controls (14 out of 15), yielding no significant difference between the two groups. Biochar treated *S. exigua* cuttings showed slightly higher survivorship (4 out of 15) than did the *S. exigua* control cuttings (2 out of 15), but was not statistically significant ($p = .41$). *S. exigua* container plants with a biochar treatment showed statistically significant ($p = .02$) higher survivorship (14 out of 15) than did *S. exigua* cuttings that also received a biochar treatment (4 out of 15). *S. exigua* container plants that did not receive a biochar treatment also showed statistically significant ($p = .002$) higher survivorship (14 out of 15) than S. exigua cuttings that did not receive a biochar treatment (2 out of 15).

Species	Plant Type	Treatment	# Planted	# Survived
P .deltoides	Containers	Biochar	15	15
P .deltoides	Containers	Control	15	15
P .deltoides	Poles	Biochar	15	13
P .deltoides	Poles	Control	15	11
S. exigua	Containers	Biochar	15	14
S. exigua	Containers	Control	15	14
S. exigua	Poles	Biochar	15	4
S. exigua	Poles	Control	15	2

Pairwise Comparisons	P-value
P. deltoides **Cont. with Biochar** vs. *P. deltoides* **Cont. without Biochar**	N/A
P. deltoides **Poles with Biochar** vs. *P. deltoides* **Poles without Biochar**	0.68
P. deltoides **Cont. with Biochar** vs. *P. deltoides* **Poles with Biochar**	0.7
P. deltoides **Cont. without Biochar** vs. *P. deltoides* **Poles without Biochar**	0.43
S. exigua **Cont. with Biochar** vs. *S. exigua* **Cont. without Biochar**	N/A
S. exigua **Poles with Biochar** vs. *S. exigua* **Poles without Biochar**	0.41
S. exigua **Cont. with Biochar** vs. *S. exigua* **Poles with Biochar**	0.02
S. exigua **Cont. without Biochar** vs. *S. exigua* **Poles without Biochar**	0.002

Table 1 – Study Site 1 - Early Spring 2017 Data

Study Site 1 – Late Summer 2017 (Cottonwoods)

Late September survivorship for the *P. deltoides* containers was 100% for both the biochar treated plantings and the controls, yielding no significant difference between the two groups. Many tags were lost during the summer due to high water levels and substantial deposition making identification of individual trees difficult, but a visual confirmation was possible to determine that every single container plant had survived. *P. deltoides* cuttings yielded no evidence of survivorship and may have suffered 100% mortality.

Figure 10 – P. deltoides cutting showing evidence of early spring growth that had dried out by the late summer data collection. No other signs of emerging vegetation were visible.

Study Site 1 – Late Summer 2017 (Willows)

I was unable to calculate the late September survivorship for the *S. exigua* container plants, largely due to the large number of tags that were lost during the summer. A visual assessment confirmed that of the 30 container plants planted, 25 survived. The 5 remaining *S. exigua* container plants that did not survive were lost to spring runoff. Even if we assume that all 15 *S. exigua* biochar treated container plants survived (15 out of 15) and that the only mortality accrued was within the control group that did not receive a biochar treatment (10 out of 15) the difference would not yield statistically significant results ($p = .32$). None of the S. exigua cuttings showed signs of vegetative growth indicating that either energy has been directed solely to the roots up to this point in time, or that there was 100% mortality among all cuttings, both those with biochar and those without, rendering any statistical analysis null and offering no significant conclusions with respect to biochar's effectiveness at assisting survival. As was the case with the *P. deltoides* plantings, some evidence may be inferred from the results confirming that *S. exigua* container plants are more likely to thrive in comparison to *S. exigua* cuttings during riparian restorations.

Figure 11 – A thriving willow container plant. A lost identification tag prevented an accurate accounting for being either a biochar treated treeor a control.

Species	Plant Type	Treatment	# Planted	# Survived
P .deltoides	Containers	Biochar	15	15
P .deltoides	Containers	Control	15	15
P .deltoides	Poles	Biochar	15	0
P .deltoides	Poles	Control	15	0
S. exigua	Containers	Biochar	15	btwn. 10-15
S. exigua	Containers	Control	15	btwn. 10-15
S. exigua	Poles	Biochar	15	0
S. exigua	Poles	Control	15	0

Table 2 – Late Summer 2017 Data

Study Site 2 – Mid Spring 2017 (Cottonwoods)

In mid-spring dormant *P. deltoides* poles that had received a biochar treatment in the late winter showed signs of higher survivorship (15 out of 15) than dormant *P. deltoides* poles that did not receive a biochar treatment (14 out of 15), but the results were not significant (p = .85).

Study Site 2 – Mid Spring 2017 (Willows)

In mid-spring dormant *S. exigua* poles that had received a biochar treatment in the late winter showed signs of higher survivorship (12 out of 15) than dormant *P. deltoides* poles that did not receive a biochar treatment (9 out of 15), but the results were not significant (p = .51).

Species	Treatment	# Planted	# Survived
P .deltoides	Biochar	15	15
P .deltoides	Control	15	14
S. exigua	Biochar	15	12
S. exigua	Control	15	9

Pairwise Comparisons	P-value
P. deltoides **with Biochar** vs. *P. deltoides* **without Biochar**	0.85
S. exigua **with Biochar** vs. *S. exigua* **without Biochar**	0.51

Table 3 – Study Site 2 – Mid Spring 2017 Data

Study Site 2 – Late Summer 2017 (Cottonwoods)

In late summer the results for dormant *P. deltoides* poles did not change; those that had received a biochar treatment showed signs of higher survivorship (15 out of 15) than dormant *P. deltoides* poles that did not receive a biochar treatment (14 out of 15), but the results were not significant ($p = .85$).

Study Site 2 Late Summer 2017 (Willows)

In late summer dormant *S. exigua* poles that had received a biochar treatment and those that did not receive a biochar treatment showed signs of equal survivorship. 2 individuals in the control group that did not show signs of survival in the early spring, did show signs of new growth by the late summer. This brought the total number of surviving control *S. exigua* poles to 11. 4 poles in each group were washed away during the spring runoff leaving 11 surviving individuals in each planting group yielding no significant results.

Species	Treatment	# Planted	# Survived
P .deltoides	Biochar	15	15
P .deltoides	Control	15	14
S. exigua	Biochar	15	11
S. exigua	Control	15	11

Pairwise Compairsons	P-value
P. deltoides **with Biochar** vs. *P. deltoides* **without Biochar**	0.85
S. exigua **with Biochar** vs. *S. exigua* **without Biochar**	N/A

Table 4 – Study Site 2 – Late Summer 2017 Data

DISCUSSION

High Mortality of Cuttings at Study Site 1

While convention dictates that a reliable evaluation of success in riparian plantings is not possible until at least 3 years after the initial planting phase (Giordanengo and Mandel 2016a), it is a distinct possibility that 100% of the *S. exigua* cuttings at Study Site 1 were lost during the summer of 2017, and it is even more likely that there was 100% mortality of the *P. deltoides* cuttings. There are several possible causes for these potential losses and one obvious one is that the cuttings themselves may have been poorly selected and may not have been appropriately evaluated for signs of disease, age, and appropriate size. For the *S. exigua* poles, which were randomly distributed among other successful container plants of the same species, there appeared to be no spatial relationship to the pattern of mortality making a stronger case for the possibility that individuals were poorly selected. Alternatively, the high mortality may have been due to how the cuttings were planted, as the job was completed by a volunteer work force, some of which were planting poles for the first time. Without the appropriate texture of the slurry backfill, *S. exigua* poles at Study Site 1 may have been at a severe disadvantage, certainly when compared to container plants that had girth in their root balls and were taller making them better able to withstand the large amount of stream deposition that occurred during the spring and summer.

Figure 12 – Left: 10% biochar soil addition being mixed in with backfill soil. Right: Volunteer work force applying backfill slurry to an S. exigua container plant. Container plants may have been better suited to withstand any insufficiently back filled planting efforts.

For the *P. deltoides* cuttings planted on the Weedy Bench area it may be less likely that the cuttings were poorly suited since they were taken from the same trees that provided thriving specimens at Study Site 2. However, spatial challenges specifically related to the depth to ground water of the area may have played a much larger issue for this group. Even the 3-foot extension of the auger used to dig the holes could not penetrate more than a few inches into the moisture below ground while convention dictates that proper installation should be total submergence of up to a few feet into the water table (Hoag 2007). A distinct line of recruitment of *P. deltoides* saplings was visible adjacent to the Weedy Bench, but the area in which the cuttings were planted was absent of any tree recruitment and instead was overrun with upland, ruderal weeds. This scarcity of wetland plants along the dry, upland berm made the location highly desirable as a potential site for a cottonwood grove, but the desiccation also made the necessity to penetrate deep into the ground water that much more imperative and unfortunately given constraints of time and money was not possible.

Figure 13 – Volunteers struggling to achieve sufficient depth for P. deltoides plantings along the Weedy Bench planting area.

Other factors that may have affected the mortality rate of the planted cuttings could have been whether and climate related. Warm, wet, autumn and winter seasons following the initial planting phase of the experiment in 2016 may have been what triggered some early budding in the non- dormant cuttings that made them vulnerable to heavy snows and colder temperatures later in the winter. The *P. deltoides* in particular were showing signs of strong survivorship in April, but the buds and remaining leaves did not look healthy, potentially indicating that they had failed to shift into dormancy and were already feeling stress before enduring what proved to be a dry, hot summer in 2017. Other successful plantings of non-dormant cuttings have received regular watering treatments (Munda et. al 2005). While this is ideal for any riparian restoration, it is often not feasible due to a lack of resources and if it is a pre-requisite it may prevent the application of non-dormant plantings at larger scales. My experiment was investigating biochar's ability to assist in water retention and therefore I administered only one additional watering shortly after the initial planting. This scarcity of additional water may have simply been too little moisture for the non-dormant cuttings to persevere in.

Long Term Survival of Container Plants at Study Site 1

Container plants are reported to be preferable to cuttings in riparian plantings when they are planted in environments where long periods of inundation and erosion are minimized, and competition within a 1- meter diameter of the plants is low to moderate (Hoag 2007). In September of 2016 competition within the Sand Bar planting area of Study Site 1 was minimal, and there was little evidence of prolonged periods of inundation, indicating good conditions for container plant survival. This may explain the high rate of survivorship for both the *P. deltoides* and the *S. exigua* container plants over the course of the first year of data collection. The zero percent survivorship of the non-dormant cuttings affirms that the type of planting stock is absolutely critical to re-vegetation efforts. While the container plants exhibited normal senescence, the non-dormant cuttings attempted to leaf out in the late fall making them vulnerable to the dry conditions that followed the initial planting phase.

Despite the strong survivorship of the container plants, their future is uncertain. The channel of the creek has shifted over the course of the season, and it is unclear whether the Sand Bar area may have to endure more substantial inundations in the future. Lack of communication between various stakeholders may have led to an unproductive planting design that funnels the stream channel water directly toward the restoration area. A riprap design on a north facing bank edge was put in place to protect a private residence that was the only structure within several hundred yards of the Study Site. Its orientation appears to have shifted the main stream channel into what was previously a backchannel. The results were disastrous for hundreds of wetland plugs that were planted in conjunction with our experiment along the moist border of the backchannel.

Figure 14 – Wetland plugs in September of 2016 (left) and September of 2017 (right). The northern perimeter of the Sand Bar planting area had almost a thousand wetland plugs planted along the back channel area. By the end of the summer in 2017 they were all washed away.

Biochar treatments were administered to some of those plugs and they were being monitored for survivorship and recruitment until the entire area was completely submerged in water, taking with it every single plug that was planted at the project's onset. The morphological shift in channel structure caused by the riprap may continue to increase periods of inundation and the intensity of stream power along the Sand Bar planting area. The container plants of both *S. exigua* and *P. deltoides* may or may not withstand this shift in the environment

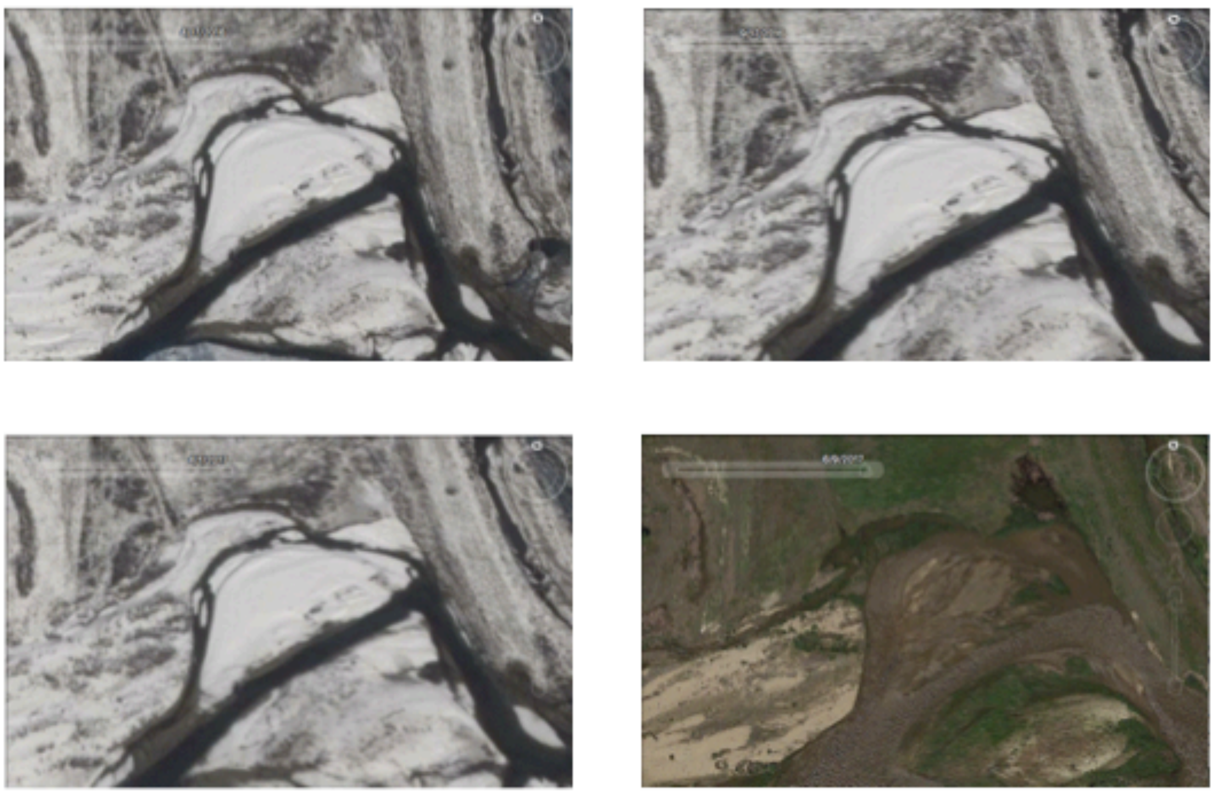

Figure 15 – Images of Study Site 1 through time generated with Google Earth Pro. Upper left is an image of the area in April of 2016, over 4 months before the project onset. Upper right is an image of the area in mid September of 2016 just before the project planting phase. Lower left is an image of the study site in April of 2017 when the spring data set was collected. Lower right is an image of the study site in June of 2017. Notice how the upper back channel has been completely blown out and the stream has engulfed the perimeter of the Sand Bar planting area. Future restructuring of the channel morphology may submerge the restoration site over time.

The Impact of Biochar at Study Site 1

While the effects of biochar on the plantings at Study Site 1 are not currently significant, it is important to note that the emerging vegetation on biochar treated plants was noticeably greater than the vegetation on the control plants. This was true for both cuttings and container plants and may indicate that biochar's potential influence was not negligible. Willow and cottonwood poles developed both leaf and buds during the extended, warm fall of 2016, and this effect was clearly more evident on poles in biochar plots. These plants may not have been phenologically adapted to subsequent freezing events. Regardless, the outcome a year later was not related to biochar amendments to soil.

Figure 16 – An early spring comparison of a P. deltoides control plant on the left, and a biochar treated P. deltoides container plant on the right. Notice the larger degree of emerging vegetation on the biochar treated sample. This appeared to be true for both the P. deltoides and the S. exigua cuttings.

I was testing for biochar's ability to help retain moisture and nutrients in dry, sandy riparian soils, and for the container plants these factors do not appear to be limiting under the current conditions. Over time, if the Sand Bar planting area is not completely washed out and inundated, moisture and nutrient limitations may begin to play a role in thinning out the planting area, and as that occurs it is possible that the biochar soil amendment will begin to play a role in determining overall survivorship. In addition, the *S. exigua* cuttings that are currently showing little to no signs of life are not necessarily going to remain in that state. It may be that biochar treated plants will be able to endure the current stress and emerge at a later date. The complex relationship between soil, climate, feedstock, and production make outcomes using biochar soils amendment difficult to predict or to replicate in any environment. However, in the case of my experiment logistical constraints made preliminary testing and investigation into the right parameters to find an appropriate combination for success even less likely and may have prevented any significant findings from emerging. It has been suggested anecdotally during conversations with industry personnel that while the fine grain texture that I used for my experiment likely retains more water due to the greater surface area, it is not as effective at maintaining the optimum conditions of appropriate soil microbes and available nutrients as is a more heterogeneous mixture that includes both fine and coarse grains. That said, the water of the St. Vrain, as measured below several towns and agricultural areas, exhibits moderate characteristics of eutrophication suggesting that more nutrients is not a concern. Preliminary greenhouse tests and ground water sampling may have helped to increase the chances of finding a better concoction than I used, but was not possible given time and budgetary limitations.

Technical and Logistical Challenges at Study Site 1

One of the most challenging aspects of the restoration was maintaining timely and fluid communication between the various stakeholders of the project. The most unfortunate result of this involved the lack of depth I achieved when planting the *P. deltoides* cuttings on the Weedy Bench. After several manual digs using a shovel and a collection of consultations with the technical adviser from WRV, it was determined that the water table on the Weedy Bench was not so deep that the 3-foot auger coupled with the 3-foot extension would not provide enough depth to reach adequately into the water table. This was, of course, not the case and it was unfortunate to discover this on the day of the project. The water table was low during the planting phase and likely declined until late April or May. Funds were available for a bobcat-attached stinger that could have potentially penetrated much deeper in to the soil and without the need for such intense manual labor, however this information was not forthcoming until very close to the project day when there was not time to address the logistical challenge of delivering the machinery to the project site. As previously mentioned the complete destruction of thousands of dollars worth of nursery-raised wetland plants was also likely the result of poor communication between stakeholders about the orientation of the riprap treatment on the stream bank and it's potential effect on the stream channel morphology. These challenges are not likely due to shortcomings in the work being done or in the preparation of achieving individual goals, but potentially speak more to the complexity of riparian restoration as a whole and the need for strong oversight and multidimensional planning efforts.

Figure 17 – Image of Study Site 1 in June 2017. Rip Rap structure is highlighted in red. The Sand Bar planting area is highlighted in purple, and the Weedy Bench planting area is highlighted in yellow. Notice the angle at which the stream flow is directed as it flows along the riprap, potentially flooding the restoration area.

Figure 18 – Additional structures that were placed along the edge of the riprap and may have contributed to the channel diversion in the direction of the restoration site are featured on the left. Their relative position along the stream is indicated on the right by the black arrows.

Another unfortunately unresolvable challenge resulted from the loss of a great number of tree tags on our planted trees. Should any significant effects from biochar emerge as the planting area thins out, they will not be able to be recorded because the biochar treated plants can no longer be identified.

Figure 19 – Willow container plants in late summer of 2017. The black arrow on the left is indicating a tag that was dragged from the base of the tree to the other limbs, presumably due to shear stress from the stream flow. The tag on the left had to be cut off in order to save the tree because it was too tight at the base. A compromise between these two extremes poses challenges without sophisticated GPS equipment.

This could have been solved through GPS technology and appropriate logging of each tree, however many GPS units are not entirely accurate to within a few meters and more expensive devices were not easily obtained. Furthermore there was substantial recruitment of both *P. deltoides* and *S. exigua* seedlings over the course of 2017, with some individuals growing nearly as large as the container plants planted along the Sand Bar. Over time, this would make the appropriate identification of trees even more difficult even with the assistance of a GPS unit, since many of these recruits are closer than a meter from trees that were planted and tagged. Willows and cottonwoods have extremely flexible stems and are designed to bend during high water flow. The tree tags I attached appeared not to have been tight enough to withstand this same degree of shear stress from the stream. If they had been attached any tighter, I believe I would have risked choking the trees to an extent that could have affected survivorship even within the first year. In fact, many of the thriving trees had already began to grow around the tags making their removal difficult or even impossible without damaging the tree.

Impressions at Study Site 2

A direct statistical comparison cannot be made with Study Site 1. These two sites differ in many characteristics not related to the focus of this study. However the high survivorship of the trees planted at Study Site 2 is not surprising given the conventional wisdom that dormant poles are preferable to non-dormant ones for riparian plantings (Giordandengo and Mandel 2016). The plantings at Study Site 2 were not intended as a restoration effort, but were merely an experiment to determine biochar's effectiveness at increasing survival. While there is currently no evidence of any effects from the biochar soil amendment, the proximity of the plantings will certainly ensure that the area will be thinned out rather quickly over time, and as that process unfolds it may be that biochar will end up playing a role in determining the survivorship of individuals. The creek at Spruce Gulch occasionally experiences episodes of periodic desiccation, and during a difficult year individuals that have received a biochar soil amendment may manage to retain more water than competing neighbors and subsequently persevere to a greater extent. Given this finding, it could prove to be useful to use biochar soil amendments for river restorations in areas wherever water and nutrients are scarce.

Closing thoughts

Despite the low success rate of the planted cuttings of both species in question, there is now a substantial population of 3rd and 4th year cottonwoods and willows on the Sand Bar area which could help stabilize the channel and expedite the return of wildlife to the area. However, an important question of interest remains; how much better off is Study Site 1 now than it would have been without the restorative effort? Only time will offer any definitive conclusions regarding this line of inquiry, but it is worth noting that recruitment of cottonwoods and willows in the area is very high, and a major restoration involving dozens of laborers takes a toll on the landscape. Over $50,000 was spent on the total restoration effort, not including the construction of the riprap wall (WRV, unpublished data). The effectiveness of the riprap aside, its presence compromised the integrity of the plantings to an extent that calls into question whether a restoration effort may have been better administered in the area at a later date. Given the lack of infrastructure in the vicinity, it is not unreasonable to postulate that the best restoration of the St. Vrain Creek along the study site area may have been to simply allow natural successional patterns to emerge and develop unaltered, and to find a healthy equilibrium in time. On the other hand, invasions from exotic plants are most manageable before they have begun. It may be that the jumpstart provided by the restoration effort served to curb any potential monocultures from developing early, in which case the time, money, and resources may have been well spent.

Figure 20 – Sand Bar planting area in the late summer of 2017.

My study initially tracked and monitored cottonwood and willow recruitment through randomly chosen grid cells, however the rebar cell divisions were washed away or buried in the spring of 2017 making it impossible to recalibrate the grid for continued assessment. If I would have found that recruitment was better in grid cells with fewer planted trees it could suggest that the best defense against exotic invaders is to allow native trees to continue to compete for resources at their own pace without intervention. At the same time the area looks as though it is more likely to attract wildlife sooner, and to provide the collection of ecosystem services that gallery forests offer to a greater extent than if the project had not been completed. The high survivorship of the container plants at Site 1 was actually superior to the rates at least two other documented restoration projects recently completed by WRV along the Front Range. A riparian section of the White Rocks Preserve in Boulder County located approximately 15 kilometers from Study Site 1 was re-vegetated with a collection of *P. deltoides* container plants and 4 different species native of shrubs two weeks after the completion of my experiment. A conservative estimate of survivorship based on recovered and lost tags yielded 43% for the container *P. deltoides*, and 62% overall for the shrubs. More liberal estimates could have recorded the overall survivorship of the container *P. deltoides* as high as 50%, but this number is still far less than what was achieved at Study Site 1. A second riparian restoration conducted in 2013 at Chico Basin Ranch located 64 kilometers southeast of Colorado Springs had less than 8% survivorship of *P. deltoides* container plants. At that site, water availability appeared to be the dominant control on survivorship. Although no statistical inference can be drawn, it appears that the restoration at Study Site 1 was highly successful and if the surviving trees manage to escape a total inundation over the next few years, the City of Longmont may have expedited the process of achieving its restoration goals. Continued monitoring of tree survivorship and the presence of target species of wildlife in the area will be essential to properly evaluating the degree of success.

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