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Natural and constructed wetlands for ecosystem and engineering services in the arid and semi-arid regions

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NATURAL AND CONSTRUCTED WETLANDS FOR ECOSYSTEM AND ENGINEERING SERVICES IN THE ARID AND SEMI-ARID REGIONS

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ABSTRACT

Natural and Constructed Wetlands for Ecosystem and Engineering Services in the Arid and Semi-Arid Regions

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The Las Vegas Wash (LVW) has undergone significant wetlands degradation and soil erosion over the past thirty years due to increasing flow resulting from urbanization and large rainfall events in the Las Vegas Valley Watershed. The increased flow and associated pollution load in the LVW and its adverse impact in Lake Mead have alerted stakeholders to pay a greater attention to explore alternative measures for rehabilitation of wetland ecosystems. This dissertation, using the case of changes in LVW, analyzes and describes ecological and engineering services provided by wetlands in arid and semi-arid regions and provides a knowledge base that can be used to improve water quality and enhance stream restoration respectively. The dissertation includes three separate studies that are organized into three independent chapters.

In the first study, constructed and naturally created wetlands in the LVW and its tributaries were studied to characterize and understand their potential role for improving ecosystem services (i.e., water purification). Excess nutrients and harmful metalloids removal was assessed at four wetlands, including Flamingo Wash Wetland, Pittman Wash Pilot Wetland, Demonstration wetlands at the city of Henderson water reclamation facility, and Las Vegas Wash Wetland. The study showed that the nutrient removal capacity of wetland vegetation in the four wetland sites correlated well with ambient nutrient concentrations in the sediments and water columns, irrespective of the type of plant present. For example, cattail and bulrush plant species have different nutrient uptake capacities, with these capacities mostly determined by the ambient nutrient and hydrologic conditions. Both species were equally efficient for nutrient uptake with high phosphorus concentration in below-ground and high nitrogen in above-ground plant parts. The below-ground parts of both species were capable of storing arsenic and selenium more efficiently than above-ground parts. However, bulrush species seem particularly efficient for removing metalloids as compared to cattail. These findings have important implications for improving our ability to engineer ecological solutions to the problem associated with common pollutants in the Las Vegas Valley.

The second project analyzed the structural and functional attributes of increasing common reeds (*Phragmites australis* (Cav.) Trin. ex Steud.) and native cattails (*Typha domingensis* Pers.) for the best ecosystem services from large scale wetlands such as the one in LVW. The entire LVW vegetation was analyzed through mapping and ground truthing to estimate areal coverage of *P. australis* Vs. *T. domingensis*. The results from this study compared with the previously published data showed that *P. australis* population is increasing in most of the places. *P. australis* in comparision to *T. domingensis*, appears to thrive better in areas with altered hydrology and high nutrient inputs. In addition to its structural dominance, our data showed that *P. australis* plays a

significant role in nutrient storage in wetlands. The net above-ground standing stock of nutrients in LVW wetlands was estimated to be approximately 26418.7 kg TN and 1264.1 kg total phosphorus (TP) for *P. australis* and 5183.8 kg total nitrogen (TN) and 272.8 kg TP for *T. domingensis*. Despite management concerns over *P. australis* dominance and growth, they fared quite well in nutrient storage in LVW wetlands compared to *T. domingensis*. The study concluded that in LVW, both *T. domingensis* and *P. australis* could be utilized for water quality improvement. It should be noted, however, plant uptake alone is not enough to improve water quality below regulatory thresholds from large scale wetlands, and managing dominant vegetation may be required for better nutrient removal efficiency.

The third project studied the riparian wetlands function for their engineering services on streambank stabilization. The mechanical properties of native species (*Artiplex lentiformis*, *Lycium andersonii*, *Larrea tridentata*, and *Allenrolfea occidentalis*) were studied to understand their suitability in revegetation purpose on banks that are easily erodible. Field experiments were conducted to estimate root length, root length density, root area ratio, and root tensile strength. Finally, the root cohesion values were assessed using a simple perpendicular model and Fiber Bundle Model. The maximum root cohesion in the present study was estimated for *A. lentiformis* (97.6kPa) followed by *L. andersonii* (89.3kPa), *L. tridentata* (35.6 kPa), and *A. occidentalis* (34.8 kPa). These values were estimated to rank the native species for their potential use in bank stabilization. The results showed that these native and most prevalent species were more suitable for shallow bank slope stabilization, since their root distributions were significantly higher in topsoil depth (0-0.5 m) in comparison to subsoil depths (>0.5 m).

This information could be utilized for revegetation and restoration purposes in the arid and semi-arid regions where these plants are abundant.

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CHAPTER 1

INTRODUCTION

1.1 Background/Problem Statement

Artificially constructed wetlands that mimic natural marshes have been used as low-cost alternatives to treat urban wastewater. Such practices have received much attention in recent years, where various aquatic plants are used for purifying the water and wastewater (Kadlec and Knight, 1996; Greenway, 2005; Thullen et al., 2005; Vymazal, 2007). The Las Vegas Valley (LVV) watershed, located in Southern Nevada, supports many ecologically significant wetlands, whereas often regarded as oases in the desert (LVWCC, 2010). As a result of increased urbanization, LVV wetlands now experience perennial surface water flows primarily comprising treated wastewater effluents with excess nutrient inputs, typical of urban influence. An important question of concern for the public and researchers is whether wetlands in the LVV have the potential to function as natural filters by improving water quality from treated wastewater effluents and urban runoff. Effluent discharges in the Las Vegas Wash (LVW) dilute water quality parameters, such as TDS, major ions, and some trace metals from urban runoff; however, effluent discharge increases nutrients, including total nitrogen (TN) and total phosphorus (TP), due to the cost associated with the wastewater treatment process. In addition, LVV is known for its elevated level of naturally occurring Selenium (Se) and Arsenic (As) in soils in some locations. The focus of the first part of the study is to compare and contrast the key characteristics of various types of wetlands to determine how well they function to remove major nutrients and toxic metals.

Like wetlands in many other rapidly growing urban areas, the LVW wetlands receive high amounts of nutrients load from treated wastewater effluents and a relatively less pollutant from nonpoint sources (LVWCC, 2009). Excessive erosion along the Wash has resulted in loss of wetlands and wildlife habitat, loss of property, damage to infrastructure, excessive sediment transport to Lake Mead, and water quality concerns in Lake Mead (LVWCC, 2000; SNWA, 2010). Restoration efforts in recent years have created more acreage of wetlands in the LVW, which is dominated by *Phragmites australis* and *Typha domingensis* species. The impact of nutrient enrichment in the wetland ecosystem may depend on how dominant wetland vegetation influences biomass production and nutrient retention (Schlesinger, 1991; Bridgham et al., 1996; Pollock et al., 1998; Grace, 1999; U.S.EPA, 2002). Very little information exists on structural and functional attributes of aquatic vegetation in response to nutrient enrichment and new management approaches in arid and semi-arid wetlands. The focus of this study is to analyze whether the *P. australis and T. domingensis* species can be utilized for the best ecosystem services in large scale wetlands.

Along with its broader ecological benefits, wetland vegetation has long been recognized by river managers for their engineering services for streambank stabilization (Thorne, 1990; Simon and Darby, 1999; Kadlec and Wallace, 2009). There is a growing recognition for important influence exerted by stream flows and channel processes on vegetation structure and composition (Pollen-Bankhead et al., 2009). Yet, there is very little information available on mechanical characteristics of xeoriparian species for bank stabilization (Simon and Collison, 2002; Simon et al., 2006; Pollen-Bankhead et al., 2009). The focus is to estimate the root cohesion values of native xeoriparian species for their possible utilization in revegetation and restoration purposes in the arid and semi-arid regions where these plants are abundant.

1.2 Research Objectives and Research Questions

- To characterize the wetland types and quantify their function for pollutants removal potential with emphasis on nutrients (TN and TP) and metals (As and Se).

  Research Question 1. How different types of wetlands in Las Vegas Valley function in removing nutrients and metals pollution? Can the wetland vegetation be managed for increasing their effectiveness?

  Hypothesis 1. Wetlands perform as a pollutant sink, where vegetation is useful for pollutant removal on both constructed and naturally created wetlands.
- To understand the structural and functional attributes of the entire Las Vegas Wash wetlands for water quality improvement in arid and semi-arid regions and compare ecosystem services between native and non-native species

  Research Question 2. Is increasing acreage of wetlands vegetation in Las Vegas Wash providing increased ecosystem services?

  Hypothesis 2. The larger the wetlands vegetation acreage the more efficient the ecosystem services are, despite vegetation types (natives or non-natives) and climatic regions (arid or humid).
- To estimate the mechanical function of native xeoriparian species to understand their suitability in revegetation for streambank stabilization.

Research Question 3. Can the xeoriparian species be utilized for restoration of stream banks and riparian ecosystems in arid and semi arid environment?

Hypothesis 3. Soil is generally strong in compression, but weak in tension. The root system is strong in tension but weak in compression. Root-permeated soil makes up a composite material that will enhance the soil strength.

This dissertation is organized in an introduction, and three manuscripts followed by a conclusion. Chapter 1 includes background, objectives, study area, and previous work. Chapter 2 contains the first manuscript describing the characteristics of various types of wetlands for pollutants removal potential. Chapter 3 is the second manuscript describing the investigation of wetland vegetation to analyze their structural and functional attributes for the best ecosystem services from large scale wetlands. Chapter 4 is the third manuscript presenting the mechanical function of riparian vegetation in streambank stabilization. Chapter 5 summarizes the conclusions from the study and outlines future work.

1.3 Study Area

The Las Vegas Valley Watershed located in Southern Nevada, an arid region of the U.S., supports many ecologically significant wetlands, which are often regarded as oases in the desert (SNWA, 2010). There are several wetlands in the LVV, some naturally formed and some constructed in various landscape positions, with a variety of hydrologic, vegetation, and soil conditions. These wetlands includes a) a constructed wastewater effluent wetland (Demonstration Wetlands at the City of Henderson Water Reclamation Facility), b) a constructed urban runoff wetlands (Pitman Wash Pilot

Wetland), c) a naturally occurring urban runoff wetland (Flamingo Wash), and d) a wetlands created by backwater behind the Pabco Road Weir in the mainstream Las Vegas Wash (LVW).

The LVW is a primary drainage channel for the 1,600 square-miles of the Las Vegas watershed that supports a substantial riparian area (Eckberg and Shanahan, 2009). In the early 1970s, the LVW channel used to be an excellent wetland habitat as the desert soil was transformed into wet marshy wetland soils. However recently**,** the LVW has experienced considerable change as a result of rapid urban development in the valley (last 20 years). The wetland areas have decreased significantly, from about 2000 acres in 1975 to about 300 acres in 1999 (LVWCC, 2000). Excessive erosion along the channel has resulted in loss of wetlands and wildlife habitat, loss of property, damage to infrastructure, excessive sediment transport and water quality concerns in Lake Mead, an artificial reservoir formed by the Colorado River (LVWCC, 2010). As a restoration initiative, many erosion controls structures are being built to stabilize the channel, lands that are adjacent to these structures are being revegetated with plants that are native to Mojave Desert riparian ecosystems. As of March 2008, 181 acres of land have been revegetated in the LVW. Also, construction of 2400 acre Nature Preserve and Wetlands Park has been initiated (Cizdziel and Zhou, 2005).

There are several wetlands along the downstream of LVW, either naturally formed as a result of flood control structures, or purposely constructed to provide ecosystem services. Wetlands along the riparian corridor of downstream Las Vegas Wash were created after the construction of erosion control structures except the Wetlands Park, which is separated from the main channel.

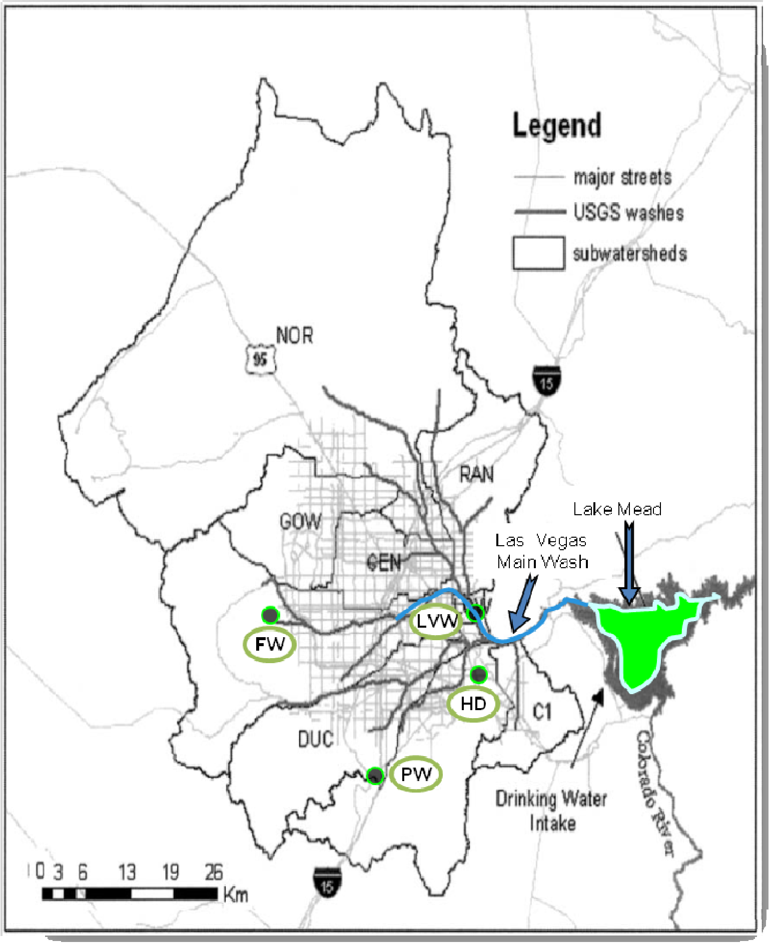

Figure 1.1 Map showing wetlands in tributaries and mainstream Wash in the Las Vegas Valley watershed (FW: Flamingo Wash, PW: Pitman Wash Pilot Wetlands, HD: Demonstration Wetland at the City of Henderson Water Reclamation Facility and LVW: Las Vegas Mainstream Wash) (adapted from Reginato and Piechota, 2004).

The wetland determination in this study was made according to the U.S. Army Corps of Engineers (Corps) wetland delineation manual (USCOE, 1987), and the procedure was adapted from the vegetation study of Eckberg and Shanahan (2009). Field experiment to study the mechanical function of xeoriparian vegetation was carried out in the Virgin River corridor, an analog site for the Lower Colorado River Basin (LCRB). Because the Wash was not historically a riverine system, it does not have an abundance of source plants native to these conditions, and plants native to Lower Colorado River Basin are being utilized for revegetation purposes in LVW wetlands.

1.4 Previous Work

1.4.1 Wetlands type and treatment function

Wetlands have been recognized for providing a higher rate of biological activity than any other ecosystems (Toet, 2003). They can transform many of the common pollutants that occur in conventional wastewater into harmless byproducts or essential nutrients that can be used for additional biological productivity (Kadlec, 1998). These biological transformations can provide an effective means to convert, release to the atmosphere, or sequester unwanted and excess chemicals from the system. Wetlands with a variety of hydrologic, vegetation, and soil conditions can occur naturally or constructed in many landscape positions (Kadlec and Wallace, 2009).

Constructed wetlands are manmade systems that have been designed to emphasize specific characteristics of wetland ecosystems for improved treatment capacity. At the current stage of technology development, three types of wetlands (free water surface-FWS, horizontal subsurface flow-HHSF, and vertical flow-VF) are in widespread use (Kadlec and Wallace, 2009). Free water surface wetlands closely mimic natural wetlands and support a wide variety of aquatic life. They are areas of open water, emergent vegetation, and designed for flow control or infiltration by the process of sedimentation, filtration, oxidation, reduction, and adsorption. The most common application for FWS wetlands is for advanced treatment of effluent from secondary or tertiary treatment processes (Vymazal, 2006). Horizontal sub surface flow wetlands consist of gravel or soil beds planted with wetland vegetation where the wastewater is kept below-ground. They are generally used of secondary treatment for small cluster systems or for small communities (Wallace and Knight, 2006). Vertical flow wetlands are designed for producing nitrified effluent and are popular in Europe. They are found to be incompatible with North American regulatory standards, which prohibit the surface exposure of fecal material (Copper et al., 1996).

The use of constructed wetlands for wastewater treatment has been tested widely in recent years, especially to reduce nitrogen (N) and phosphorus (P) loads (Vymazal, 2006). Phosphorus in wetlands occurs as phosphate in organic and inorganic compounds. Free orthophosphate is the only form of P believed to be utilized directly by algae and macrophytes, thus represents a major link between organic and inorganic P cycling in wetlands (Toet, 2003; Reddy et al., 2005). Phosphorus transformations in wetlands include adsorption, desorption, precipitation, dissolution, plant and microbial uptake, fragmentation, leaching, mineralization, sedimentation, and burial (Kadlec, 1999, 2005). Phosphorus storage in vegetation can range from short to long-term, depending on the type of vegetation, litter decomposition rate, leaching of P from detritus tissue, and translocation of P from above to below-ground biomass (Reddy et al., 2005). Phosphorus storage in above-ground biomass of emergent macrophytes is usually short-term, with a large amount of P being released during the decomposition of litter. Thus, the above-ground portions of macrophyte return P to the water, while a below-ground portion returns P to the soil (Tanner, 2001).

Similarly, N compounds are among the principal constituents of concern in wastewater because of their role in eutrophication, and their toxicity to aquatic life (Kadlec and Wallace, 2009). These compounds also augment plant growth, which in turn stimulates the biogeochemical cycles of the wetlands. The processes that affect removal and retention of N during wastewater treatment in wetlands include NH_3 volatilization, nitrification, denitrification, N fixation, plant and microbial uptake (assimilation) (Vymazal, 2007). Nitrogen assimilation is one of the major transformations and refers to a variety of biological processes that convert inorganic N forms into organic compounds and serve as building blocks for cells and tissues. The potential rate of nutrient uptake by a plant is limited by its net productivity (growth rate), the concentration of nutrients in the plant tissue (Wetzel, 2001), and on the ultimate potential for biomass accumulation (Richardson and Vymazal, 2000). Therefore, desirable traits of a plant used for nutrient assimilation and storage would include rapid growth, high tissue nutrient content, and the capability to attain a high standing crop (Reddy and DeBusk, 1987).

Research findings from wetland bioassessment suggest that trace amounts of metals have been reported in plants growing in natural and constructed wetlands for wastewater treatment (Lesage et al., 2007; Vymazal and Krasa, 2005; Vymazal et al., 2009). Wetlands are found to be effective at retaining significant loads of toxic metals primarily in wetland sediments. There is also a greater concern toward the cumulative loading of some trace metals, which might have reverse impact on the aquatic biota (Hamilton, 2004; Lin and Terry, 2003).

Several attempts have been made to develop and adopt advance technologies for treatment wetlands (McBride and Tanner, 2000; Langergraber, 2005; Rousseau et al., 2005; Wu and Huang, 2006; Vymazal et al., 2007; Kadlec and Wallace, 2009). Treatment performance is represented by two components, the central treatment tendency for a wetland (or a group of wetland) and the anticipated variability away from that central tendency. Central tendencies are driven by flows and concentrations, in concert with environmental factors. Treatment performance of wetland systems are being studied by several comparative studies including inflow and outflow concentration of water and sediments, hydrological setting, retention time, and seasonal differences etc. (Moore et al., 1994; Kadlec, 1999; U.S. EPA, 2000; Mitsch et al., 2004). There have been several comparative studies to elucidate possible effects of vegetation type, media size, and physiochemical parameters (Theis and Young, 2000). The result showed that aggregated data sets on those variables can be used to best define the central tendency in treatment performance of each wetland system. The graphical representation of treatment performance essentially extends the idea of percent removal to a group of wetlands, but it is more realistic once the information on the detention time or hydraulic loading is included (Hammer and Knight, 1994; Vymazal, 2001; Knight et al., 2004).

Like wetlands in many other rapidly growing urban areas, the LVV wetlands receive relatively high amounts of nutrients from wastewater effluents and potential pollutants from nonpoint sources. Water quality in the LVV wetlands is mainly determined by the effluents from three wastewater treatment facilities. Effluent discharges dilute water quality parameters, such as TDS, major ions, and some trace metals (including Se) from urban runoff; however, effluent discharge increases nutrients, including N and P, as a result of the wastewater treatment process. Another emerging issue is naturally occurring trace metals, for e.g. higher concentration of Se and As is reported in some locations of Las Vegas Valley. Flows from the tributaries are the major sources of contaminants and have particular concern to the LVV wetlands, mainly As and Se. With the historically observed wetland area, there is a growing concern over bioaccumulation of these trace metals and pollutants.

1.4.2 Wetland vegetation: structure and function

Vegetation-based indicators can be utilized to determine whether ecological integrity has been impaired by nutrient enrichment in naturally created wetlands (USEPA, 2002). The growth and reproduction of vegetation as well as large scale primary production are frequently limited by supplies of N or P in a freshwater ecosystem (Elser et al., 2007). These nutrients are also responsible for changes in ecosystem function and structure that occur when wetland assimilative capacity is exceeded (Carpenter et al., 1998). The structural attributes include characteristics of the community or of individual species, where the changes occur through shifts in plant species composition (Craft et al., 1995; Bridgham et al., 1996). It includes the replacement of nutrient intolerant native species by exotic species usually adapted to high nutrient conditions. The functional attributes related to energy flow and nutrient cycling, where the changes occur in response to nutrients include increased N and P uptake, NPP, and decomposition (Davis, 1991; USEPA, 2002). The understanding of wetland structure and function for water purification and pollutant removal has been of a great interest to researchers since 2000 (Pu et al., 1998; Hu et al., 1998; Mitsch et al., 2001, 2005; Jiang et al., 2007; Mander and Mitsch, 2009; Wang et al., 2009; Zhang et al., 2009), as aquatic vegetation has a great ability for assimilating large amounts of nutrients from sediment and overlying water during the growing season.

There is considerable information on N and P concentration in plant tissue as well as standing stocks for plants found in natural and constructed wetlands (Vymazal et al., 1998, 2009; Tanner, 1996, 2001; Mitsch and Gosselink, 2000; Greenway, 2005; and Kadlec, 1999). Researchers argue that nutrient removal can be optimized by selecting suitable species with higher capacity for N and P absorption and conversion into plant biomass (Mitsch and Gosselink, 2000; Greenway, 2003; Vymazal, 2007). However, large difference in effluent improvement exists between different plant species (Tanner, 1996; Karathanasis et al., 2003; Iamchaturapatr et al., 2007).

An understanding of plant biomass and nutrient uptake is essential to characterize the ecosystem function (Mayer and Edwards, 1990). It has been found that the performance efficiencies of constructed or natural wetlands depend on several variables, such as the quality and quantity of effluent to be treated, biological, physical, and chemical activities in that particular wetland system (Greenway and Woolley, 2001; Greenway, 2003). The TN and TP content of living biomass in wetland vegetation varies considerably among species, among plant parts, and among wetland sites. Treatment wetlands are often nutrient-enriched and display higher values of tissue nutrient concentrations than naturally occurring wetlands. Large differences in N and P content among different plant parts is found to be the result of translocation, seasonality, and genotypical habit (Cronk and Fennessy, 2001). Emergent aquatic plant species such as cattails (*Typha*), bulrushes (*Schoenoplectus*), and reed (*Phragmites)* have been widely used in the U.S. and elsewhere around the world for nutrient removal in constructed wetlands (Kadlec and Wallace, 2009; Tanner, 2001) . The compartments analyzed in plant tissues are centered within live above-ground plant tissues, where as the below-ground parts are usually not considered or often omitted. In treatment wetlands and naturally existing wetlands that are lightly loaded and covered a relatively small surface area, below-ground storage may be an important factor in the nutrient dynamics (Prentki et al., 1978).

Wetlands may also release large amounts of nutrients during decomposition and water treatment function shift toward the carbon cycling and denitrification process (Thullen et al., 2005; Chimney and Pietro, 2006). It has been reported that system functioning benefits from higher plant diversity and improvements in plant selection and cultivation may facilitate nutrient removal from wastewater (Engelhardt and Ritchie, 2001). Most constructed wetlands are low in plant diversity or even monocultures, and one attempt to improve the role of plants in constructed wetlands was to increase the plant diversity (Zhang et al., 2007; Brisson and Chazarenc, 2008). Karathanasis et al. (2003) and Amon et al. (2007) have reported the higher efficiency of mixed wetlands for effective root distribution, less susceptible to seasonal variations, and had more diverse microbial populations than monoculture wetlands. According to Engelhardt and Ritchie (2001), management practices that maintain the diversity of aquatic macrophytes in wetlands, by sustaining or restoring a natural disturbance regime to prohibit exclusion of less competitive species, may sustain ecosystem function and services of wetland. Changes in species composition, loss of overall plant diversity, conversion of a unique flora to one dominated by a few common species, and replacement of native species by exotics have been reported in connection with nutrient enrichment in wetland ecosystems (Koerselman et al., 1990; Ehrenfeld and Schneider, 1991; Thullen et al., 2005).

The LVW supported around 2000 acres of wetlands until 1975, when the base flow discharge was small (Alcorn, 1988). After the rapid urbanization in Las Vegas Valley, the increasing wastewater flow resulted into extensive soil erosion and lateral saturation has been reduced, thereby resulting in wetland degradation (SNWA, 2010). Restoration efforts have been initiated since the year 2000 to protect the LVW, which includes construction of bend weirs, bank stabilization, and revegetation of native species. Vegetation monitoring in LVW is being conducted by Las Vegas Wash Coordination Committee (LVWCC) in the revegetation sites basically to ensure the compliance with the requirement set by federal and state funding agencies. However, the structural and functional attributes of wetland vegetation in the LVW has not been assessed cumulatively for the best ecosystem services from large scale wetlands such as the one in LVW.

1.4.3 Riparian vegetation function for bank stabilization

In addition to the ecological benefits of wetlands, engineering services provided by root networks of riparian vegetation act to increase the apparent cohesion of soil through a combination of mechanical and hydrologic effect (Pollen and Simon, 2005). The riparian fluvial system in the semi-arid region is characterized by very less precipitation throughout the year. It often experienced the cycles of flash floods causing the substantial channel widening followed by channel scouring and soil erosion (Osterkamp and Costa, 1987). Such flooding events might change the meandering channels into wide braided morphology, which creates more space for vegetation growth toward the braided island and floodplain banks (Bankhead et al., 2009). Together with bank stabilization structures, vegetation is widely believed to increase the stability of streambank (Simon and Collision, 2002). The stabilization effect of vegetation on soils is currently the subject of field and experimental studies attempting to explain channel stabilization, morphology and patterns in fluvial systems over a wide range of temporal and spatial scales (Gran and Paola, 2001; Pollen and Simon, 2005). Revegetation efforts are well established as an effective means of restoration for erosion control measure (Baets et al., 2007). Revegetation strategies for erosion control focused on both above-ground and below-ground effects of vegetation where the above-ground biomass will help on resettling the sediments and reduce the water flow. The below-ground vegetation plays an important role for root reinforcement through root tensile strength, and matric suction through evapotranspiration (Simon and Collison, 2002). However, the quantification of these reinforcing effects will require the detail investigation of root systems, and the impact of environmental variability on root architecture.

Many subsequent descriptions of root systems in arid environments have been studied by previous researchers. According to Rundel and Nobel (1991) and Gibbens and Lenz (2001), the strong spatial and temporal heterogeneity of soil water in deserts has resulted in very divergent patterns of rooting architecture. Besides the uptake and storage of water and nutrients, another principal role of root is the provision of stability for the plant itself, implying resistance against wind, water, gravitational forces, and for the soil containing the roots (Reubens et al., 2007). The soil adjacent to the roots is affected both hydrologically and mechanically, in terms of aggregate stability, infiltration capacity, soil bulk density, soil texture, organic and chemical content, and shear strength (Morgan, 2005; Reubens et al., 2007). Detailed studies from laboratory and the field have examined the effects of roots on erosion during concentrated overland flows (Gysesels and poesen, 2003; Gyssels et al., 2005; De Baets et al., 2006; De Baets et al., 2007; Hubble et al., 2010; Pollen and Simon, 2010; Schwarz et al., 2010), and observed exponential decline rates of soil detachment with increasing root length densities and root biomass. The soil environmental characteristics have a major influence on root system development and ultimately determine the effectiveness of roots for soil fixation. According to De Baets et al. (2006) root architecture plays an important role in the reduction of soil erosion, with fine roots are shown to be particularly effective at preventing soil detachment. The relationship between soil material movement and root structure are mainly based on bulk root characteristics such as root density (RD, the dry mass of the living roots per unit soil volume), root length density (RLD, the total length of the living roots per unit soil volume), and root area ration (RAR, the root cross sectional area per unit soil surface) (Smit et al., 2000; Reubens et al., 2007). However, the understanding on root architecture of xeoriparian shrubs and its application in the stabilization effect has been less studied.

The important mechanical feature of roots is that they are strong in tension, on the other hand, soils are strong in compression and weak in tension (Simon and Collison, 2002; De Baets et al., 2008). A combined effect of soil roots results in a reinforced soil and magnitude of such reinforcement depends on root distribution and root tensile strengths (Greenway, 1987; Gray and Barker, 2004; De Baets et al., 2008). Root tensile strength (in situ root pull-out test and laboratory root tensile test) revealed that the number and morphology of root system influences the stress-strain relationship and ultimate resistance to failure (Riestenbreg, 1994; Abernethy and Rutherford, 2001; Schmidt et al., 2001; Bischetti et al., 2005; Norris, 2005; Pollen and Simon, 2005; Tosi, 2007; Comino and Marengo, 2010). Root tensile strength decreases with increasing root diameter by following a power law relationship (Operstein and Frydman, 2000; Tosi, 2007; Mattia, 2005; Bischetti et al., 2005; De Baets et al., 2008; Comino and Marengo, 2010). The interspecies differences in tensile strength are less significant to bank stability than the interspecies differences in root distribution (Abernethy and Rutherford, 2001). The fine roots are also known for higher tensile strength in comparison to coarse roots and contribute more to soil reinforcement (Operstein and Frydman, 2000; Tosi, 2007; De Baets et al., 2008). Several authors have quantified the shear strength of soils, both with and without roots, and found that root increases the soil shear strength, normal components of soil resistance, and modify the shear zone width (Waldron, 1977; Wu et al., 1979; Waldron and Dakessian, 1981; Abe and Ziemer, 1991).

The processes of soil reinforcement by roots can be considered as an example of the recently developed ecological concept of ecological engineering. Initial attempts to quantify root reinforcement of soil have been dominated by the use of simple perpendicular root models developed by Waldron (1977) and Wu et al. (1979). This model requires the tensile strength of the roots, and the cross-sectional area of root system crossing the shear plane. Wu’s model estimate maximum root reinforcement at a single instance of time, since all of the roots available in the soil matrix have reached their maximum tensile strength (Pollen and Simon, 2005). Also the Wu’s model overestimates the reinforcement values because it assumes that all roots crossing the shear plane break at the same time, and they further propose a fiber bundle model (FBM).

The FBM assumes that roots within the soil matrix have different maximum strengths, and therefore break at different points as a load is applied to the soil. Also, this model redistributes the load from the broken roots to the remaining intact roots crossing the shear surface (Pollen and Simon, 2005). Root cohesion values are broadly estimated and applied for soil stabilization by several researchers in the past decade (Abernethy and Rutherford, 2001; Simon and Collison, 2002; Pollen and Simon, 2005; Tosi et al., 2007; De Baets et al., 2008; Bischetti et al., 2009; Preti and Giadrossich, 2009; Hubble et al., 2010; Comino and Druetta, 2010; Comino and Marengo, 2010). Many studies have estimated root cohesion for a variety of riparian tree and herbaceous species (Abernethy and Rutherfurd, 2001; Simon and Collison, 2002; Pollen and Simon, 2005; Simon et al., 2006; Pollen-Bankhead et al., 2009), but substantially less work has been carried out to study the root cohesion values of riparian shrub species for streambank stabilization.

Riparian ecosystems support many critically important ecological functions within western landscapes (Brinson et al., 1981), but riparian areas have been severely degraded by the detrimental effects of flow regulation, overgrazing of rangelands, mining, and urbanization (Follstad Shah, 2007). Restoration of riparian ecosystem has become a major enterprise across the U.S. Southwest since 1990 to achieve a goal of reversing degradation (Goodwin et al., 1997; Stromberg, 2001). The modifications of stream flow by dams and diversions in the Colorado River have significantly affected the riverine marshlands. As stream flows become more intermittent, wetland vegetation reduced and species composition in the floodplain shifts from wetland pioneer trees (*Populus fremontii* , *Salix gooddingii*) to more drought tolerant exotic shrubs including *Tamarix ramosissima* and *Pulchea sericea* (Ohmart et al., 1988). Restoration efforts are underway in U.S. Southwest with special focus on re-establishing hydrogeomorphic processes through restoring appropriate flows and manipulating vegetation structure by planting native species (Stromberg, 2001; Stromberg et al., 2007). River managers are widely applying native species revegetation to increase streambank stability among other purposes (Simon and Collison, 2002). Although the effects of vegetation on bank stabilization are broadly explored, native shrubs from desert ecosystems are rarely quantified for their engineering services in bank stabilization.

1.5 Summary

Several investigations have pursued treatment wetlands as a low-cost solution for improving water quality in rapidly growing urban areas. In this study, the key characteristics of constructed and naturally created wetlands in the Las Vegas Valley watershed will be determined to understand their function to improve water quality. The wetland vegetation (one species of cattail and three species of bulrush) will be investigated for their potential in nutrient (N and P) and trace metal (Se and As) uptake among four different wetlands. Plant tissue concentration will be compared among four wetland sites to understand nutrients and toxic metals storage potential with respect to the ambient concentration in the water column and sediment. By understanding the limits of wetland function, watershed management actions can be tailored to improve ecological services in the Las Vegas Valley.

From the previous studies, it is obvious that the structural and functional attributes of wetland vegetation can be utilized for the optimum treatment purposes. Structural attributes of wetland vegetation will be analyzed to estimate plant frequency, cover, and species distribution, followed by vegetation mapping and ground truthing. Functional attributes will be analyzed through nutrient storage potential in dominant wetland vegetation i.e. *T. domingensis* Vs *P. australis*. The focus will be to understand whether exotic *P. australis* and native *T. domingensis* can be utilized for the best ecosystem services from large scale wetlands in arid and semi-arid regions.

Wetlands vegetation has long been recognized for its broader engineering services including streambank stabilization. The desert native shrubs *A. lentiformis* (Torr.) S. Watson (Quail bush), *L. andersonii* A. Gray (Wolfberry), *L. tridentata* (DC.) Coville (Creosote bush), and *A. occidentalis* (S. Watson) Kuntze (Iodine bush) will be studied to understand their suitability in revegetation for bank stabilization. The root cohesion values will be estimated applying simple perpendicular model and recent fiber bundle model. The root cohesion values will be beneficial for ranking species in revegetation purpose in the Lower Colorado River Basin.

The dissertation consists of an introduction, three manuscripts including removal of nutrients and metals by constructed and naturally created wetlands in the Las Vegas Valley, vegetation assessment for the nutrient uptake potential of macrophytes in semi-arid wetlands, and estimation of root cohesion for desert shrub species in riparian ecosystem of arid and semi-arid regions and its potential for streambank stabilization, followed by a general conclusion section.

CHAPTER 2

REMOVAL OF NUTRIENTS AND METALS BY CONSTRUCTED AND NATURALLY CREATED WETLANDS IN THE LAS VEGAS VALLEY, NEVADA

Increased water use associated with rapid growth in the Las Vegas Valley has inadvertently led to the creation of unique wetland systems in Southern Nevada with an abundance of biological diversity. Constructed and naturally created wetlands in the Las Vegas Valley watershed were studied to characterize and understand their potential role for improving ecosystem services (i.e., water purification). Nutrient and metal removal was assessed at four sites including a natural urban runoff wetland, a constructed urban runoff wetland, a constructed wastewater wetland, and a natural urban runoff/wastewater wetland. Plant nutrient uptake was dependent on ambient nutrient concentrations in water and sediments of specific wetlands, irrespective of the type of plants present. Phosphorus was mostly concentrated in below-ground plant parts whereas nitrogen was concentrated in above-ground parts. As for metalloids, bulrushes were more efficient than cattails at taking up arsenic and selenium. Averaging all the wetland sites and plant species, total nitrogen, phosphorus, arsenic and selenium removal was 924.2, 61.5, 0.30, and 0.38 kg/ha/yr, respectively. Present findings suggest that natural and created wetland systems can improve water quality in the Las Vegas Valley watershed for some common pollutants, however, other measures are still needed to improve water quality below regulatory thresholds.

2.1 Introduction

Wetlands with a variety of hydrologic, vegetation, and soil conditions can occur naturally or be constructed in many landscape positions (Kadlec and Wallace, 2009). Wetlands are often highly productive systems where numerous biological transformations are taking place, driven by the natural energies of the sun, soil, wind, and by microorganisms, plants, and animals (Thullen et al., 2005). Performance efficiencies of constructed or natural wetlands depend on several variables, such as the quality and quantity of effluent to be treated, and biological, physical, and chemical activities in that particular wetland system (Greenway and Woolley, 2001; Greenway, 2003). Until recently, nitrogen and phosphorus were primary constituents of concern in wetland systems, with their concentrations varying depending on the source of wastewater and the extent of nonpoint source pollution (Vymazal, 2006; Toet et al., 2005). However, recently other pollutants, such as heavy metals, radioactive chemicals, and pharmaceutical and industrial organic chemicals have also emerged as pollutants of concern.

Wetland plants mediate important processes in constructed wastewater treatment wetlands. For example, plant metabolic activity releases oxygen into the rhizosphere, which aids in nitrification through the direct uptake of nutrients (Brix, 1997; Greenway and Woolley, 2001). The access and availability of nutrients affects plant growth response and resource allocation, which influences removal efficiency in wetlands (Tanner, 2001). Emergent aquatic plant species such as cattails (*Typha* spp.), bulrushes (*Schoenoplectus* spp.), and reed (*Phragmites australis*) have been widely used in the U.S.

and elsewhere around the world for nutrient removal in constructed wetlands (Kadlec and Wallace, 2009). Nutrient removal can be optimized by selecting suitable species with higher capacities for absorption of inorganic nitrogen and phosphorus and conversion into plant biomass (Greenway, 2003; Vymazal, 2007; Mitsch and Gosselink, 2000). A basic understanding of the growth requirements and characteristics of wetland plants is essential for successful design and operation of wastewater treatment. Several authors have studied the importance of vegetation in removing metals from natural and constructed wetlands for wastewater treatment (Lesage et al., 2007; Vymazal and Krása, 2005; Vymazal, 2007). Bioaccumulation processes are found to be effective in reducing some metals such as arsenic (As) and selenium (Se) into insoluble forms in some constructed wetlands (Zhang and Moore, 1997; Zhang and Frankenberger, 2003; Lin and Terry, 2003).

The Las Vegas Valley watershed located in Southern Nevada, an arid region of the U.S., supports many ecologically significant wetlands and is often regarded as an oasis in the desert (LVWCC, 2009). Excessive erosion has resulted in the loss of wetlands and wildlife habitat, loss of property, damage to infrastructure, excessive sediment transport, and water quality concerns in Lake Mead (LVWCC, 2009). Wetlands have decreased significantly, from about 2000 acres in 1975 to about 300 acres in 1999 (Eckberg and Shanahan, 2009). The multi-stakeholder Las Vegas Wash Coordination Committee developed a management and enhancement plan to restore the ecological services of the Las Vegas Valley's primary drainage channel, the Las Vegas Wash. As a restoration initiative, many erosion control structures are being built to stabilize the channel and lands that are adjacent to these structures are being revegetated with plants native to Mojave Desert riparian ecosystems. Like wetlands in many other rapidly growing urban centers, the wetlands in Las Vegas receive relatively high amounts of nutrients from wastewater discharge and potential pollutants from nonpoint sources. For example, selenium concentrations in urban runoff channels in the Las Vegas Valley are above regulatory thresholds. Consequently, wetlands have been pursued as a low-cost solution for improving water quality in various locations in the valley. Until now, performance of these wetlands has not been cumulatively assessed.

The goal of this study was to compare and contrast the key characteristics of various types of wetlands in the Las Vegas Valley watershed to determine how well they function to improve water quality. The nutrient (nitrogen and phosphorus) and trace metal (selenium and arsenic) uptake by wetlands plants (one species of cattail and three species of bulrush) in four different wetlands was investigated. Above-ground and below-ground plant parts were compared between each site to understand nutrient and metalloid uptake and storage with respect to the ambient concentration in the water column and sediment. By determining the limits of wetland function, watershed management actions can be tailored to improve ecological services in the Las Vegas Valley.

2.2 Material and Methods

2.2.1 Study Area

The study was carried out in four lowland wetlands types (elevation less than 2,100 feet) in the Las Vegas Valley (Figure 2.1), including a) a constructed wastewater effluent treatment wetland (Demonstration Wetlands at the City of Henderson Water Reclamation Facility, ‘HD’ hereafter), b) a constructed urban runoff treatment wetland

(Pitman Wash Pilot Wetland, ‘PW’ hereafter), c) a naturally occurring in-situ urban runoff treatment wetland (Flamingo Wash, ‘FW’ hereafter), and d) a natural wetland created behind an erosion control structure in the main Las Vegas Wash (Las Vegas Wash, ‘LVW’ hereafter).

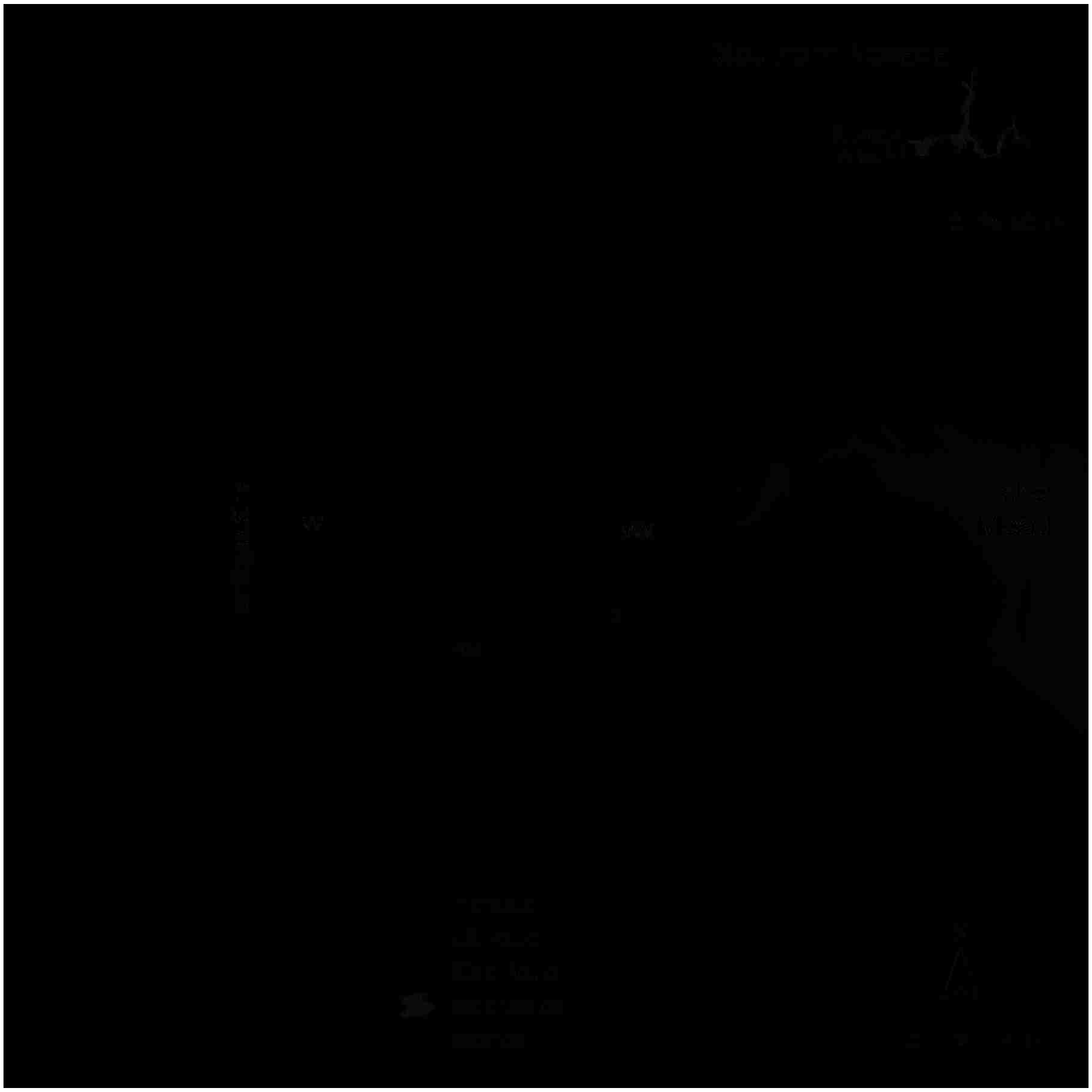

Figure 2.1 Map showing different wetlands sites located within the Las Vegas Valley Watershed (FW: Flamingo Wash, PW: Pitman Wash Pilot Wetlands, HD: Demonstration Wetland at the City of Henderson Water Reclamation Facility, and LVW: Las Vegas Wash).

The Las Vegas Valley is a low-lying alluvium-filled valley surrounded by steep mountain ranges. Soil cover in the study area generally consists of depositional silts and clays from the Quaternary era. Intermittent streams continue to cut into the floodplain and deposit alluvium into the surrounding wetlands.

Las Vegas Wash Wetlands (LVW): The Las Vegas Wash (36^{0}06'49.23" N and 115^{0}08'53.17" W) is the major drainage for the Las Vegas Valley, which drains into Las Vegas Bay in Lake Mead. The Las Vegas Wash currently discharges ~290 cubic feet per second (cfs) providing nearly 2% of the inflow to Lake Mead (Leising, 2003; SNWA, 2010; USGS, 2010). The Las Vegas Wash wetlands site, which consists mostly of treated wastewater effluent from three municipal facilities, is located in the main channel of the Las Vegas Wash and was created from the backwater pool behind the Pabco Road erosion control structure (i.e., weir). The LVW meets stringent water quality standards set by Nevada Division of Environmental Protection at all times for the safe return of water to Lake Mead and Colorado River. Land use type around the LVW wetlands are dominated by undeveloped desert areas and mixed riparian vegetation. The wetlands area extends nearly 220 acres and the wetland vegetation in this area is dominated by cattail (*Typha domingensis*) and common reed (*Phragmites australis*). The Las Vegas Wash also conveys untreated urban runoff, groundwater, and stormwater (Zhou et al., 2004).

Flamingo Wash Wetlands (FW): These wetlands are located in the Flamingo Wash (36^{0}05'17.02" N, and 114^{0}59'10.80" W), a tributary to the Las Vegas Wash, and consist of urban runoff with an average discharge of ~5 cfs. The adjacent lands are dominated by dense residential, commercial, and park/golf course uses. The Flamingo Wash stretches for several miles but the wetlands are somewhat patchy and sparsely located (~5 acres).

Dense vegetation of annual weeds mixed with cattails exists throughout the channel and provides habitat to many aquatic and avian species.

Pittman Wash Pilot Wetlands (PW): The Pittman Wash ($36^0$04'31.79" N and $115^0$00'07.07" W) is a demonstration-type pilot wetland created to study water quality improvements in urban runoff before it enters the Las Vegas Wash. The PW wetlands are experimental (20 m by 20 m) and have both surface and sub-surface flow components and a discharge of ~5 cfs and total area of 0.009 acres. The surrounding land use type is similar to that of the FW. The main vegetation in the PW wetlands is three species of bulrushes (*Schoenoplectus acutus, S. americanus,* and *S. californicus*).

Demonstration Wetlands at the City of Henderson Water Reclamation Facility (HD): This is another demonstration-type wetlands located at the City of Henderson Water Reclamation Facility ($36^0$02'48.29"N and $115^0$03'13.06" W). This site was constructed to show how wetlands can improve partially treated wastewater effluent. The land use type consists of residential and undeveloped land. The 5.75 acre wetland is a triangular-shaped pond with 14 loafing and emergent vegetation islands constructed with varying depths of water coverage. Three species of bulrush (*S. acutus, S. americanus,* and *S. californicus*) were planted on eleven specially designed hummocks.

2.2.2 Sampling and Analyses

2.2.2.1 Water

Water samples were collected monthly from all four sites from inlets and outlets beginning in July 2008 and ending in June 2009. Various parameters, including total nitrogen (TN, measured as NO_3+NO_2+NH_4), total phosphorus (TP, measured as orthophosphate), dissolved oxygen (DO), pH, electrical conductance, and temperature, were measured from the four sites. Nalgene bottles (1 liter) used during sampling were acid rinsed prior to the sampling. Water samples were then immediately stored on ice. TP concentration was determined using the colorimetric analysis after persulfate digestion (APHA, 2005). TN was analyzed using an automated colorimetric method using a Lachat QC8000. Metal analysis of water samples were determined by ICP-MS using a method based on USEPA Method 200.8 (USEPA, 1991).

2.2.2.2 Sediment

Sediment samples were collected from the same inlet and outlet locations as the water samples at all four wetlands seasonally. Vertically mixed sediment samples were collected using a plastic scoop up to ~10 cm depth and transferred into 100 ml glass bottles with polyvinyl caps. Samples were then dried in a convection oven at 70^{o}C until they were completely dry. Subsamples of dry sediment (~1 g) were processed for metal digestion following USEPA Method 3050B at the Desert Research Institute Ecological Engineering Laboratory. Sediment samples were digested with repeated addition of 70% HNO_3 and 30% H_2O_2. A low-temperature thermostat (Lauda Ecoline, U.S. version) was used to provide uniform heating of 95^{o}C. The resultant digest was diluted to 100 ml, centrifuged, and stored at 4^{o}C until analysis. Samples were analyzed for trace metals using inductively coupled plasma optical emission spectrometry (ICP-OES) at the Goldwater Environmental Laboratory at Arizona State University. Sediment TP content was analyzed for 1 g dry subsamples using the colorimetric method (APHA, 2005).

Sediment TN content was analyzed on a dry subsample (~1 g) using a PerkinElmer 2400 CHN analyzer.

2.2.2.3 Plant

Plant samples were collected seasonally between the inlet and outlet locations of all four wetlands using 0.5 m by 0.5 m quadrants. A total of 14 quadrants were selected, 5 in LVW wetlands and 3 each in HD, PW, and FW wetlands for vegetation study and sampling purposes. The LVW wetland was sampled at five quadrants due to its larger size compared to the rest. All plant material (above- and below-ground) in each quadrant was harvested and measured for biomass, nutrients (TN and TP), and suite of metals. Plant biomass was calculated using methods described in APHA (2005) for dry plant weight by storing for 72 hours at 70^{o}C or until a consistent dry weight was obtained. Dry plant samples were separated into roots, stems, and leaves prior to sub-sampling for nutrients and metals analyses. A Cyclone Sample Mill (UDY Corporation, Fort Collins, Colorado) was used to grind dry plant tissue to a homogenate sample of approximately 1 mm in size for nutrient and metal analyses. Plant TP and TN contents were determined using the methods used for sediment analyses. For metals, 1 g plant samples were digested following USEPA Method 3050B. Digested samples were processed for metal concentration using ICP-OES. Twenty-nine trace metals were analyzed in plant, sediment, and water samples. Among the detected metals, selenium and arsenic were critically evaluated because of their higher concentrations, known presence in the valley, and potential adverse impact on water quality and aquatic wildlife. QA/QC protocols were based on standard methods and included reagent blanks, check standards, fortified samples, laboratory and field duplicates and certified reference materials for water, sediment and plant samples (APHA, 2005). All samples were analyzed at EPA certified laboratory.

2.2.3 Statistical Analysis

Statistical analyses were carried out using JMP software (SAS Institute, Cary, North Carolina). One-way analysis of variance (ANOVA) was used to study the effect of wetlands type and plant species on the nutrient and metal concentrations in plants. Two-way ANOVA was used to study the interactions of wetlands type and species distribution with TP, TN, and metal concentrations. Differences detected in ANOVAs from the wetlands sites were compared using the Tukey pairwise comparison test. For all of the tests, p-values < 0.05 (95% confidence interval) were considered significant. Plant, water, and sediment nutrients and metals were regressed among sites to see correlations among them.

2.3 Results

Water quality parameters other than nutrients and metals such as TSS, BOD, pH, temp etc were generally consistent in all the wetlands. The treatment facilities are fitted with tertiary treatment systems and do a good job of keeping the TSS and BOD low in the LVW wetland similar to urban and residential runoff fed wetlands (FW, HD and PW wetlands). On average, pH and temperature range from 7.2-8.1 and 22 to 25^0C at all four wetland sites. Similarly, average DO and TSS range between 6-10 mg/L and 4 to 47 mg/L in all the wetlands.

2.3.1 Plant Biomass

Most of the plants in the LVW and FW wetlands were cattails, whereas the HD and PW wetlands were dominated by three species of bulrush. The total mass of cattails and bulrushes varied significantly among the four wetlands sites (Table 2.1). Of the two cattail dominated sites, LVW had a greater average biomass production than FW (Table 2.1). For the bulrush sites, all three bulrush species had higher biomass in HD than in PW. Overall, total biomass harvested per quadrant was highest in the HD wetlands compared to the other three wetlands (Table 2.1).

Table 2.1 Average biomass and nutrient concentrations of above-ground plant parts of *Typha domingensis* and *Schoenoplectus* spp. at the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetlands at the City of Henderson Water Reclamation Facility (HD), and Pittman Wash Pilot Wetlands (PW). Digits after ± sign indicate standard errors.

Site	Species	Total Culm per Quadrant	Total Biomass per Culm	Total Biomass	TN Storage	TP Storage
		(number)	(kg)	(kg/m^2)	(g/m^2)	(g/m^2)
LVW	*T. domingensis*	14 ± 5	0.27±0.05	9.69±0.21	135.7±12	6.6±0.6
HD	*S. americanus*	17 ± 6	0.26±0.04	11.37±0.17	152.4±9.3	16.0±1.0
	S. californicus	13 ± 5	0.35±0.07	11.20±0.29	170.2±17.8	13.4±1.4
	S. acutus	15 ± 6	0.11±0.04	4.09±0.15	48.3±6.9	4.7±0.7
PW	*S. americanus*	11 ± 4	0.16±0.05	4.61±0.19	44.7±7.2	2.2±0.4
	S. californicus	14 ± 9	0.16±0.03	3.79±0.13	37.5±5.4	1.5±0.2
	S. acutus	14 ± 9	0.11±0.03	2.26±0.11	15.8±3.0	0.5±0.1
FW	*T. domingensis*	11 ± 3	0.08±0.03	2.62±0.12	28.6±5.2	1.5±0.3

2.3.2 Nutrients Analysis

Plant, sediment, and water column nutrient data measured at the various wetlands differed in mean concentrations ($p<0.05$). Annual average plant tissue analyses indicate that TP concentration varied significantly among the four wetlands ($p<0.05$), showing that TP was significantly different among the HD, PW, and FW wetlands for both cattail and bulrush plants ($p < 0.05$). TP concentration in the LVW wetlands, however, was similar to that in FW ($p = 0.55$, Figure 2.2a). Plant tissue %TP generally followed the trend of the ambient sediment and water column concentrations for the wetlands sites rather than for the individual species. The HD wetlands had the highest average sediment TP concentration (0.08%), followed by the LVW (~0.045%), PW (~0.043%), and FW (~0.03%) wetlands. The pairwise comparison showed that sediment TP concentration in the HD wetlands was significantly different than in the LVW and FW wetlands (Tukey LSD, Figure 2.2b). Unlike in plants and sediments, phosphorus concentrations in the water were not significantly different among the PW, FW, and LVW wetlands. However, the HD wetlands had a significantly higher TP concentration, ~ 1.5 mg/L, in the water column (Tukey LSD, Figure 2.2c). Overall, the annual mean TP water concentrations were ~ 0.145 mg/L at the LVW, ~ 0.01 mg/L at FW, and ~ 0.010 mg/L at the PW wetlands.

All four wetlands had significant drops in sediment TP concentrations at the outlets ($p < 0.01$). A relatively lower reduction of 16% was measured at LVW, whereas the reduction was nearly 60% at the FW, 30% at the PW, and 26% at the HD wetlands. Unlike sediment concentrations, there was no significant decrease in water TP concentrations toward the outlets. From regression analysis, plant tissue TP concentrations were found to be highly correlated with sediment concentrations ($R^2= 0.83$, Figure 2.3a) and moderately significant at 90% confidence level ($p< 0.1$). The annual average phosphorus concentrations in the water column were also positively correlated with plant tissue concentrations among the four wetland sites at 90% confidence level ($R^2= 0.85$, $p < 0.1$, Figure 2.3b).

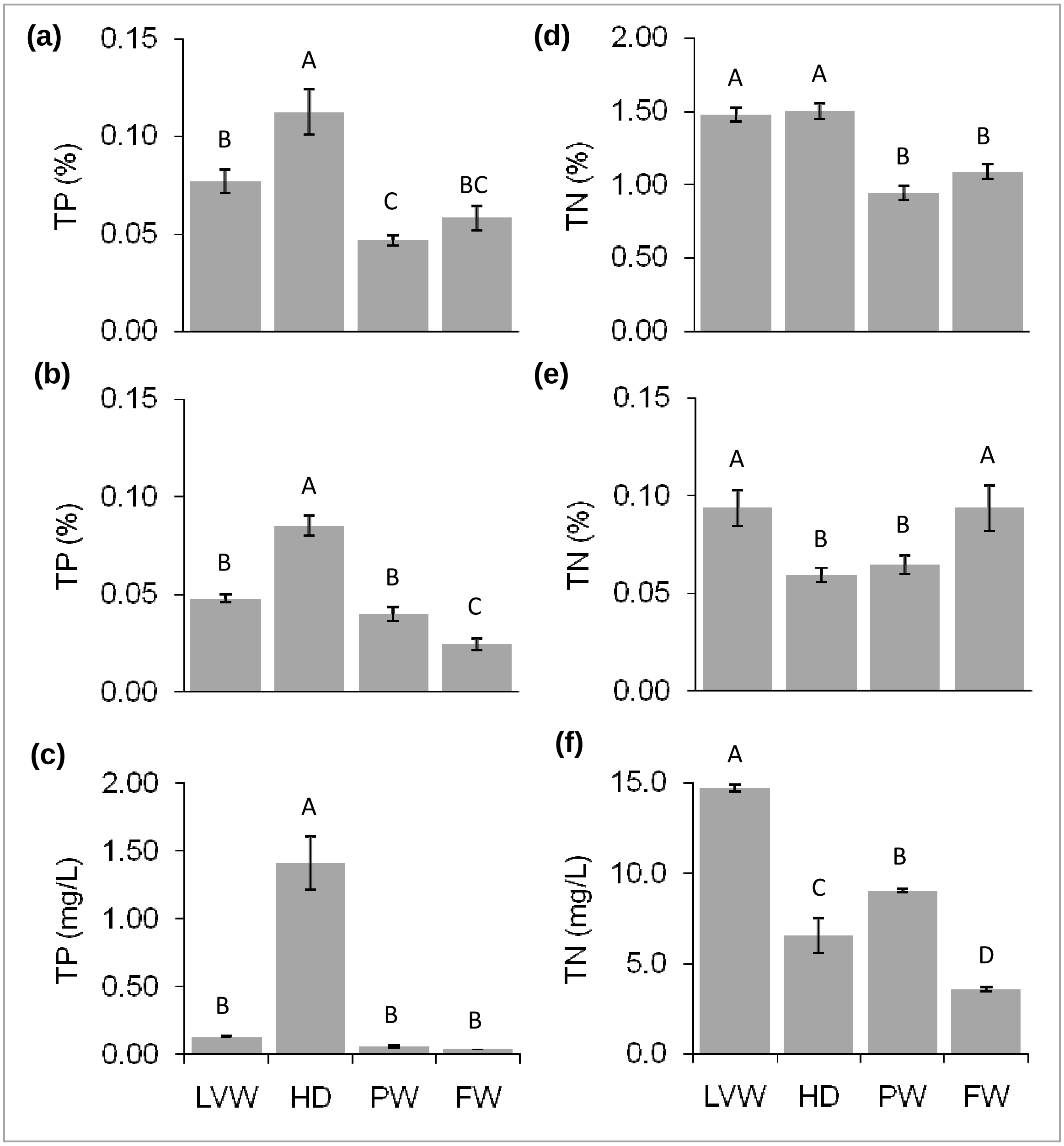

Figure 2.2 Average annual total phosphorus (TP) and total nitrogen (TN) concentrations in a, d) plants (*Typha domingensis* and *Schoenoplectus spp.*); b, e) sediments; and c, f) water at the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetlands at the City of Henderson Water Reclamation Facility (HD), and Pittman Wash Pilot Wetlands (PW). Letters above bars denote significant differences based on pairwise (Tukey HSD) comparisons. Error bars represent standard errors.

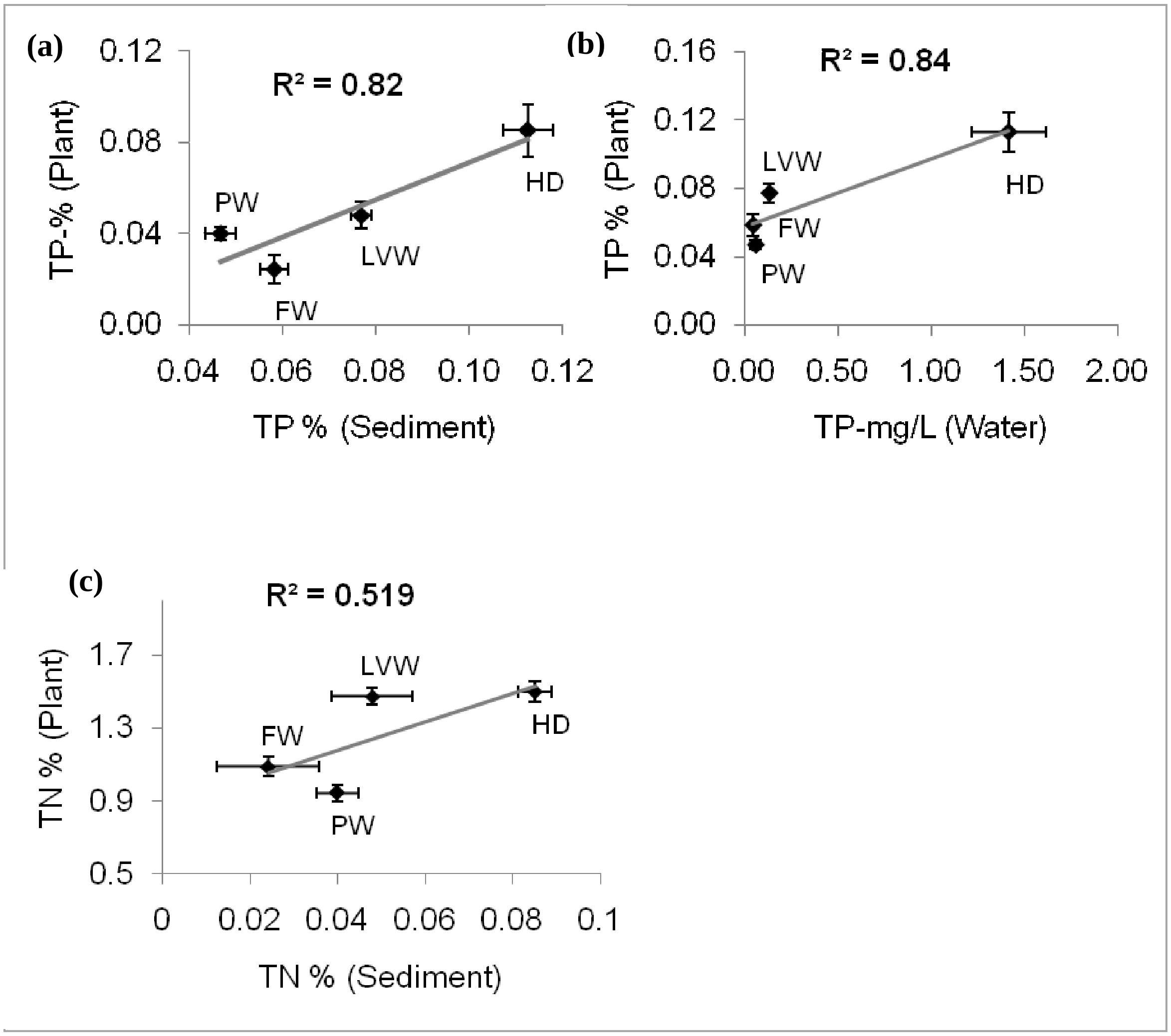

Figure 2.3 Overall correlations between annual average plant tissue and a) sediment total phosphorus concentrations (TP%), b) water column total phosphorus (mg/L), and c) sediment total nitrogen (TN%) in the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetlands at the City of Henderson Water Reclamation Facility (HD), and Pittman Wash Pilot Wetlands (PW). The line shown is a least square linear regression.

TN concentrations measured in cattail and bulrush plants were significantly different among the four wetlands ($p<0.05$). Cattail plants in the LVW wetlands and bulrush in the HD wetlands appeared more efficient in N storage compared to the other two wetlands (Table 2.1, Figure 2.2d). As for the sediment nitrogen, LVW and FW had the highest TN concentration (0.09%), followed by PW (0.06%), and HD (0.05%). Pairwise comparisons showed that sediment TN in the HD and PW wetlands was significantly different from sediment in the LVW and FW wetlands (Tukey LSD, Figure 2.2e). Nitrogen concentrations in the water columns were also significantly different among the four wetlands ($p < 0.01$, Figure 2.2f). Overall, the mean TN concentration in water at the LVW wetlands (14 mg/L) was higher than in the PW wetlands (~9 mg/L) and FW wetlands (~4 mg/L). Despite consisting of only treated wastewater effluent, the inlet of the HD wetlands had a lower mean TN (~ 5 mg/L) than the inlets of the LVW and PW wetlands. There was a significant drop in sediment %TN at the outlets ($p< 0.01$). This reduction of TN in FW was 61%, followed by 23% for HD. The other two wetlands (PW and LVW) had less than 5% reductions. Average TN concentrations in water measured at the inlet and outlet of the LVW wetlands did not show any major differences. The regression analysis did not reveal any correlation between the plant tissue TN concentration and the water column TN concentration. However, the plant tissue TN concentration was moderately correlated to the sediment TN concentration among the four wetland sites ($R^2= 0.51$, $p<0.1$) at 90% significant level (Figure 2.3c). Above-ground plant parts for both species were more efficient at taking up TN at all four wetlands when compared to below-ground parts ($p< 0.01$), whereas below-ground plant were more efficient for TP uptake (Tukey LSD, Figure 2.4a, b).

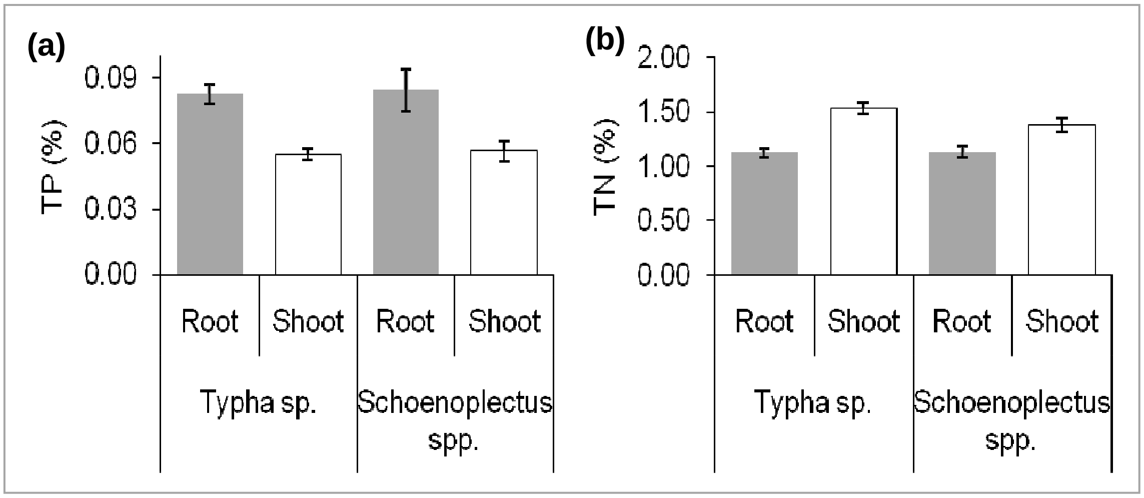

Figure 2.4 Average annual a) total phosphorus (TP), b) total nitrogen (TN) in the shoot and root parts of plant tissues (*Typha domingensis* and *Schoenoplectus spp.*) at the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetlands at the City of Henderson Water Reclamation Facility (HD), and Pittman Wash Pilot Wetlands (PW). Error bars represent standard errors.

2.3.3 Metals Analysis

Among a suite of trace elements analyzed, As and Se were detected at relatively higher concentrations at all wetlands sites and were studied in more detail due to their history in the Las Vegas Valley watershed. Several other trace metals e.g., Hg, Pb, Zn, Cd, Fe, and Mo in plants and Cd, Co, Cr, Cu, Hg, Fe, Li, Ni, Pb, and Zn in sediments were detected, but all were under the MCL (maximum contaminant level) (USEPA, 2004). The concentrations of these metals showed no significant differences among four wetland sites.

Among the four wetlands, the PW wetlands had the highest average annual As concentration in plants, sediments, and water. PW plants (bulrushes) had ~6.0 µg/g As, which was significantly higher than the As levels in the other wetland sites ($p < 0.01$,

Figure 2.5a). LVW plants (cattails) had the second highest As concentration (~3.5 µg/g). However, the tissue concentrations of As were relatively lower in FW and HD wetland plants. Similarly, annual mean sediment As concentrations were significantly different among the four wetlands sites ($p < 0.01$, Figure 2.5b). Also, sediment in the PW wetlands had the highest concentration (~6.06 µg/g) followed by LVW (~4.71 µg/g), FW (~3.65 µg/g), and HD (~3.36 µg/g). Similar to the plants and sediments, the water column As concentrations differed among the four wetland sites ($p< 0.01$, Figure 2.5c). There was no significant decrease in As concentrations in sediment from inlet to outlet in any of the wetland sites. The PW wetlands had the highest concentration of As (13.1 µg/L) in the water column, followed by LVW (~7.1 µg/L), FW (~4.47 µg/L), and HD (~3.42 µg/L). Generally, As concentrations in the water column at the outflow sites were similar to those at the inflow sites and did not show any significant reduction. Regression analysis showed that the annual average As concentrations in plant tissues were highly correlated with the sediment concentrations at 90% confidence level ($R^2= 0.98$, $p< 0.1$, Figure 2.6a) and water column concentrations ($R^2= 0.88$, $p< 0.1$, Figure 2.6b) among the four wetland sites.

There was a remarkably high Se concentration (~9.80 µg/L) detected in the bulrush plant tissues in the PW wetlands. The rest of the wetlands each had about one-fourth of the concentration of Se as in the PW wetlands. Cattails appeared to have lower Se concentrations at both the LVW (~2.32 µg/L) and FW wetlands (~1.29 µg/L) as compared to the bulrushes of the HD (2.5 µg/L) and PW (9.8 µg/L) wetlands (Figure 2.5d). The LVW and FW wetlands sediments measured higher concentrations than the HD and PW wetlands ($p< 0.01$, Figure 2.5e).

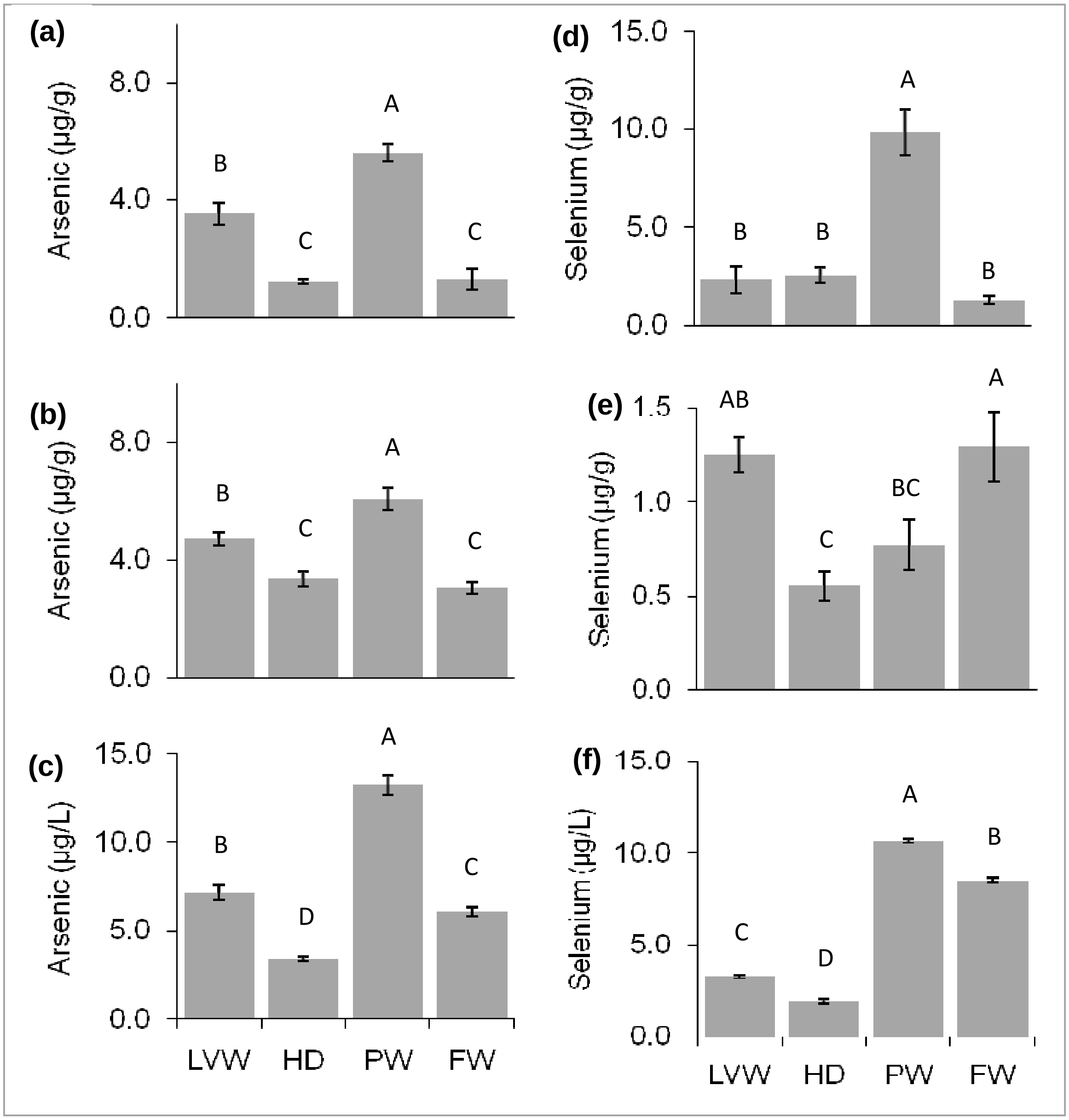

Figure 2.5 Average annual arsenic(As) and selenium(Se) concentrations in, a & d) plants (*Typha domingensis* and *Schoenoplectus spp.*); b & e) sediments; and c & f) water at Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetlands at the City of Henderson Water Reclamation Facility (HD), and Pittman Wash Pilot Wetlands (PW). Letters above bars denote significant differences based on pairwise (Tukey HSD) comparisons. Error bars represent standard errors.

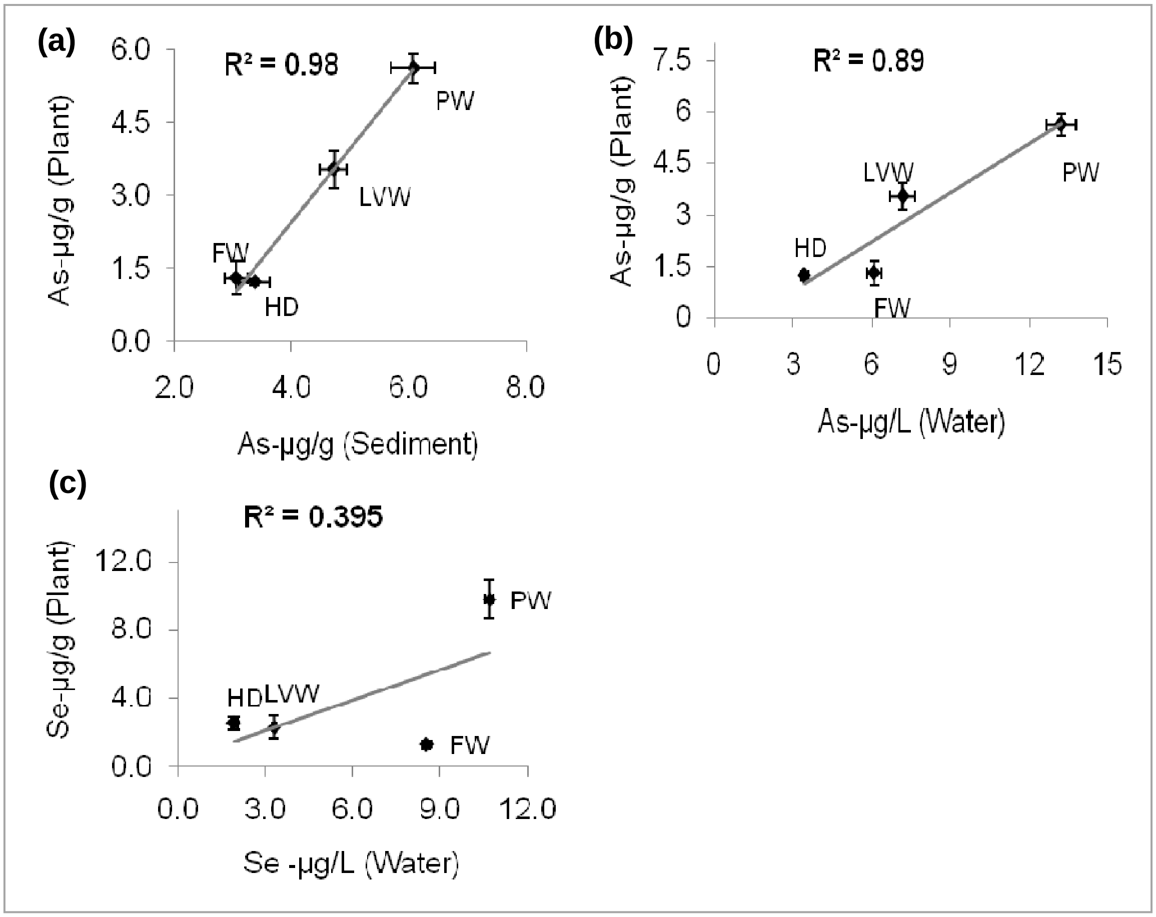

Figure 2.6 Overall correlations between annual average plant tissue (µg/g) and a) sediment (µg/g), b) water arsenic (As) concentrations (µg/L), and c) water selenium (Se) concentrations (µg/L) in the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetlands at the City of Henderson Water Reclamation Facility (HD), and Pittman Wash Pilot Wetlands (PW). The line shown is a least square linear regression.

The annual mean sediment Se concentrations were higher in FW (1.3 µg/g) and LVW (1.2 µg/g) but relatively lower in PW (~0.77 µg/g) and HD (~0.55 µg/g). Annual average Se concentrations in the water column were significantly different among the four wetland sites ($p< 0.01$, Figure 2.5f). The PW wetlands had the highest concentration of Se in the water column (~10.68 µg/L), followed by FW (~8.2 µg/L), LVW (~3.2

μg/L), and HD (~1.91 μg/L). Se concentrations in sediment did not show any significant differences between the inlets and the outlets among the four wetland sites. Similarly, As concentrations in the water column at the outflow sites did not show any significant reductions. Regression analysis was not significant between plant tissue and sediment Se concentrations. However, the plant tissue Se concentration was weakly correlated with the water column concentration among the four wetland sites ($R^2 = 0.39$, $p < 0.1$, Figure 2.6c). At all sites, comparing above-ground and below-ground data revealed that Se and As concentrations were significantly higher in the below-ground parts of either species than in the above-ground parts ($p<0.05$, Figure 2.7a, b).

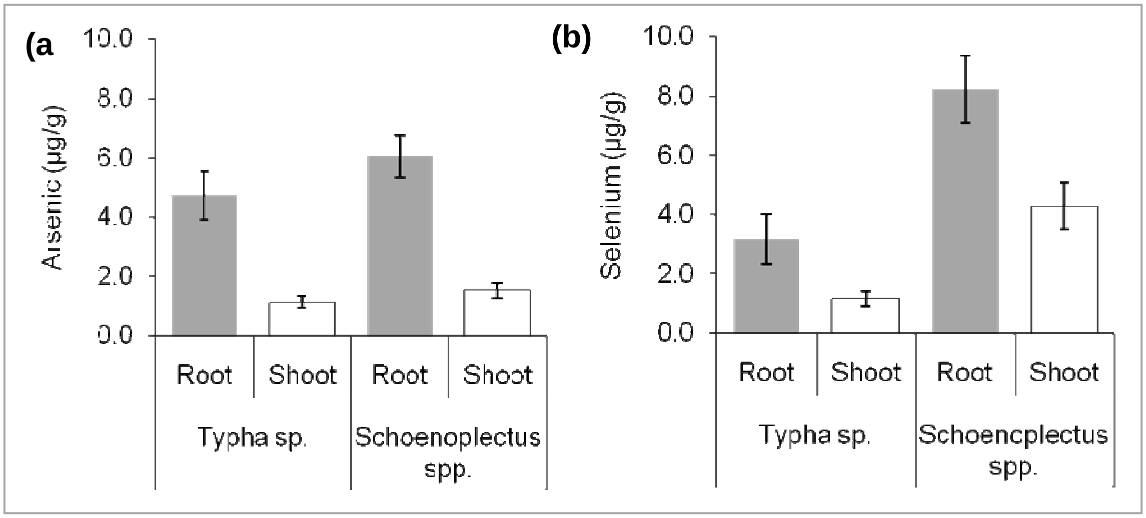

Figure 2.7 Average annual a) arsenic (As) and b) selenium (Se) concentrations in the shoot and root parts of plant tissue (*Typha domingensis* and *Schoenoplectus spp.*) at the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetlands at the City of Henderson Water Reclamation Facility (HD), and Pittman Wash Pilot Wetlands (PW). Error bars represent standard errors.

2.4 Discussion

2.4.1 Plant Biomass

Cattail and bulrush biomass ranged from 2.2-11.3 kg/m^2/yr, which is comparable with constructed wetlands in highly productive ecosystems. Total plant productivity at the end of the vegetation cycle was estimated to be 13-20 kg/m^2/yr for cattails and bulrush species in constructed ecosystems but was only 3-5 kg/m^2/yr in natural and less-polluted areas (Vymazal et al., 1998; Mitsch and Gosselink, 2000; Reddy and De Busk, 1987). In this study the peak standing crop was measured, which is also known as the single largest value of plant material present during a year's growth (Richardson and Vymazal, 2000). Plant productivity and nutrient accumulation in plant biomass varied widely for cattail and bulrush species among the four different wetland sites (Table 2.1). This variation could be due to differences in environmental parameters such as incoming nutrients and hydrology in the wetland systems. For example, bulrushes, especially *S. americanus*, showed a high density of stem growth in the HD wetlands but relatively less density and biomass in the PW wetlands.

Similarly, cattails in the LVW wetlands yielded higher plant density and biomass per quadrant compared to the FW wetlands. The LVW and HD wetlands receive high nutrient loads from wastewater treatment plants, whereas the PW and FW wetlands receive relatively lower nutrient loads as they are fed by urban runoff systems. In both of these cases, incoming nutrients might have played a role in the plant densities. Aquatic plants take up large quantities of nutrients and assimilate them efficiently (Cronk and Fennessy, 2001). The present results show that the plants may be capable of growing better by taking up more nutrients (if available in the wetlands system) and producing more biomass. The biomass values measured in this study represent maximum seasonal biomass values and are higher than productivity estimates that include a carryover of biomass from the previous season. For HD, the restrictive nature of hummocks and multi seasonal growth might be the major reasons behind high plant biomass. Because the exact age of the plants was not known, some plants might represent two or more growing seasons. However, plants representing two growing seasons do not necessarily carry maximum nutrient concentrations (Reddy and De Busk, 1987).

2.4.2 Nutrients Analysis

Present study data suggest that nutrient concentrations tended to be highest for *S. californicus* compared to the other two bulrush species. Cattails were also found to have relatively higher nutrient concentrations. Cattail plants in present wetland sites had high nutrient uptake compared with similarly constructed wetlands in other parts of the U.S (Kadlec and Wallace, 2009). In a study by USEPA (2000), two free-water surface treatment cells at the Iron Bridge Wetland in Florida, *S. californicus,* and *T. latifolia* removed TN and TP to a similar extent. Nitrogen uptake by cattails and bulrushes was in the range of 100-300 g N/ m^2 at different constructed treatment wetlands in the U.S. (Kadlec and Wallace, 2009); this is comparable to the present results. However, the nutrient storage per m^2 in this study differs significantly because of the plant biomass values varying among the four wetlands (Table 2.1). High densities of bulrush species carried large amount of nutrients in the system, up to 170.2 g TN/m^2 and 16.0 g TP/m^2. Nutrient storage results are on the high end compared to the findings of Vymazal (2006), who reported that the nitrogen standing stock for emergent species was in the range of 14 to 156 g N/m^2. Similarly, Tanner (2001) showed that bulrush plant tissues accumulated 8.8-13.4 g TP/m^2 and 48-69g TN/m^2 in total biomass (root and shoot). These data are within a close range of present studied wetlands systems.

TN and TP contents of living biomass in different wetlands vary considerably among species, plant parts, and wetland sites (Table 2.1). Despite their differences in total biomass, nutrient concentrations in plant tissues were similar between cattail and bulrush species. Nutrient content per unit of biomass was generally more site-specific than species-specific. This is not unique only to the present system; for example, another study found that nutrient removal efficiency of a system depends on the plant type, growth rate, nutrient composition of the water, and physicochemical environment in the water-sediment system (Reddy and De Busk, 1987). Also, in the present study, below-ground parts appear to be more efficient in phosphorus uptake compared to the above-ground plant parts (of both cattails and bulrushes). However, in contrast, above-ground plant parts had higher nitrogen concentrations compared to the below-ground parts for both species. These results are in agreement with Greenway (2005), who compared nitrogen and phosphorus in root/rhizomes and leaf/stem tissues for a variety of native wetlands species in constructed wetlands in Queensland, Australia, and found that the nitrogen content was highest in the above-ground parts and the phosphorus was highest in the below-ground parts.

Species differences had little to no affect on TP uptake, rather the ambient concentration of nutrients in the sediments appeared to drive differences among the specific wetland sites. Sites with higher ambient nutrients also had generally higher nutrients in the plants. This is not completely unexpected because plants have higher plasticity for nutrients. This has also been found in many algal nutrient studies; for example, a previous study found that algae grown in higher nutrient concentrations have higher algal N and P concentrations due to weaker homeostasis in plants compared to other organisms (Acharya et al., 2004; Sterner and Elser, 2002). There was a noticeable reduction between the inlet and outlet sediment TP concentration for all the wetlands. However, reductions were less significant and highly variable for TP in water. Phosphorus removal in the water column is highly variable and depends on many factors such as settling of fine particles, among others. This is also suggested in a study by Kadlec and Wallace (2009) of 250 different free-water surface wetlands that showed that the reduction of phosphorus from inflow to outflow is unpredictable and variable.

Also, nutrient data in this study suggested that TP concentrations in plant tissue had relatively higher correlation with concentrations in the sediments and water columns (Figure 2.3a, b). Relative concentrations were particularly strong in the HD wetlands (Figure 2.2a, b, c). This is perhaps expected considering that the HD wetland receives treated effluent from a wastewater treatment plant and the wetland has a long retention time. Similarly, other previous studies suggest for TP removal contact time may play a major role in the distribution within constructed wetlands (Drizo et al., 2000), and it has been suggested that the removal efficiency of TP is positively correlated with retention time (Klomjek and Nitisoravut, 2005).

Total nitrogen measured in water and sediments were higher in the LVW wetlands than in other wetlands (Figure 2.2e, f). The source of the higher nitrogen input (~14 mg/L) is the effluent coming from the wastewater discharge (~290 cfs) in the LVW wetlands. Whereas the FW wetland, which is a tributary of the LVW wetland, receives much less discharge (~ 5 cfs) and has much less nitrogen in the system show higher difference in removal(between inlet and outlet concentrations). Both of these wetlands have similar hydrology and plant types. Comparing the difference between inlet and outlet measurements, the FW wetlands were found to be more efficient in sediment nutrient removal. Higher discharge might be too much to overcome for the wetlands in LVW to substantially increase removal of nitrogen from the system. Despite a loss in TN at the outlet of the FW wetlands, cattails in the FW wetlands generally had lower TN concentrations than in the LVW wetlands. This may be due to a less favorable habitat for plants to flourish in channel wetlands combined with other means or nitrogen removal such as denitrification. Furthermore, nutrient inputs can directly modify or change biological communities. Fluctuations in hydrological conditions induce changes in nutrient inputs. Therefore, high dependence on hydrology is particularly important in semi-arid and arid areas, where surface water levels fluctuate seasonally (Mitsch and Gosselink, 2000).

In this study, nitrogen uptake by plants was not significantly correlated with either ambient water and sediment concentrations, as suggested by the weak regression coefficients for both the water column and sediment (Figure 2.3c). Different hydrological regimes observed in LVW wetlands might have contributed to different TN and TP concentrations in the plants, sediments, and water columns. Despite less nitrogen input and lower water and sediment concentrations, TN recovery through plant assimilation was remarkably high in the HD wetlands as compared to PW wetlands. It may be due to the denitrifying of pond water by the City of Henderson Water Reclamation Facility in March 2008 just prior to the sampling date. Also, the plants have been growing in the HD wetlands for several years and thus were growing when TN concentrations in the water and sediment were much higher than during this study period (Zhou and Van Dooremolen, 2007). Better performance of the HD wetlands might also be due to the better vegetation management practice of using hummocks. A study in the southwestern U.S. found the properly configured hummocks in constructed wastewater treatment wetlands can be used to maintain the proper balance of vegetation necessary to optimize treatment function (Thullen et al., 2005).

2.4.3 Metals Analysis

Similar to nutrients, both cattail and bulrush species were effective bioaccumulators of these metalloid pollutants (As and Se) from the wetland systems. This study suggested that As and Se uptake capacity was significantly higher in bulrushes than in cattails. Among the three species of bulrush, *S. americanus* was the most effective at As and Se uptake, followed by *S. acutus* and *S. californicus*. However, both of these latter species are also known to acquire heavy metals in their root, rhizome, and leaf tissues, as found in studies of constructed wetlands for treatment of pond effluents in Alabama, U.S. (Schwartz and Boyd, 1995) and for metal contaminated urban streams in southeast Queensland, Australia (Cardwell et al., 2002). Similarly, below-ground plant tissues (root) had higher concentrations of both As and Se than the above-ground (shoot) parts (Figures 2.7a, b). Present findings are comparable with the study by Vymazal et al. (2009), who found that concentrations decreased in the order of roots > rhizomes > leaf > stems for 19 different trace elements, including As and Se, for *Phragmites australis* plants growing in constructed wetlands with subsurface flow for treatment of municipal sewage in the Czech Republic.

A number of trace metals are essential micronutrients at low concentrations, but some trace metals may occur in wastewater at concentrations that are toxic to aquatic wildlife (Hamilton, 2004; Fox and Doner, 2003). Concentrations of As in the plants of the four wetlands were consistent with the trends in the ambient concentrations of the sediments and water columns (Figure 2.5a, b, c). The regression analysis showed that the As in plants is significantly correlated with sediment and water column (Figure 2.6a, b) concentrations. Overall, the highest measured As uptake in plants was in the PW wetlands followed by the LVW wetlands. The HD and FW wetlands had the lowest plant As concentrations. Among the four wetland sites, the PW wetlands also had the highest sediment and water column As concentrations, followed by LVW wetlands. The As concentration (13.12 µg/L) measured in the water column of the PW wetland exceeds the drinking water standard (10 µg/L). It is thought that the As is naturally found in the soils of the Las Vegas Valley and enters the Wash and its tributaries through shallow groundwater discharge rather than from anthropogenic sources (Cizdziel and Zhou, 2005). Sediment from the outlets of the PW and HD wetlands showed a small drop in As concentrations, but this was not the case in the FW and LVW wetlands. In contrast to As, among the four wetlands, Se concentrations in plants were relatively more consistent with water column than with sediment concentrations. Se concentrations in the sediments of the four wetlands (<2.0 µg/g) were moderate and perhaps without any consequential impact on aquatic life (Figure 2.5 e). A Se concentration of less than 2 µg/g is considered below the toxicity threshold (USEPA, 2004). Unlike concentrations in the sediments, Se concentrations in the water column were relatively higher (10-15 µg/L) in both the FW and PW wetlands. Regression analysis between plant tissue Se concentrations and Se in the water columns among four wetland sites (Figure 2.6c) was relatively weak (than As).

Waterborne Se concentrations, of FW (8.2 µg/L) and PW (10.68 µg/L) wetlands exceeded the EPA standard for chronic exposure (5 µg/L) and even came close to acute exposure (20 µg/L) (USEPA, 2004). Although fish and wildlife may be exposed to an elevated risk of Se toxicity, the site specific evidence provided by Hamilton (2004) showed the risk level is low to moderate for our studied wetlands. Se concentrations analyzed in plants from the LVW, FW, and HD wetlands (>3.0 µg/g) are similar to those found in the study by Seiler et al. (2003) in the western U.S. The results for plant Se concentrations in the LVW, FW, and HD wetlands are similar to those of Pollard et al. (2007) for bulrushes and cattails in the Nature Preserve wetlands and Hansen et al. (1998) for shoot and root tissues of wetland plants in the constructed wetlands of the San Francisco Bay. Seiler et al. (2003) provided a typical background level for plant tissue Se (1.5µg/g) and dietary effect levels in these tissues (~3 µg/g). Se concentration in plants from LVW, HD, and FW were below these levels and only plant tissues in PW exceeded (~10 µg/g) this level. The PW wetlands' relatively high Se concentrations could pose an elevated risk of bioaccumulation for birds and wildlife and transfer to higher trophic levels in the food chain. Se concentration in the PW wetlands in bulrush plant tissue, sediments, and water column is similar to Se concentrations in constructed wetlands from other parts of the world (Kadlec and Wallace, 2009). Kadlec and Wallace's study compiled Se concentrations in vegetation in treatment wetlands exposed to Se, and found that they were typically in the range of 1-20 µg/g for plants and 1-10 µg/g for sediments.

2.4.4 Seasonal Variation

Seasonal variation of nutrients (TN and TP) and metals (As and Se) were also analyzed to see whether there were any noticeable differences in storage potential for specific plant and wetland type. There were only a few signals of variations but these trends were not validated by statistical testing (Appendix A-XIV). For example, seasonal average TP concentrations in the cattails were higher in LVW during the summer season but there was no apparent difference between spring and winter. Similar trends were seen at HD and PW wetlands for TP% in bulrush plants with typically higher concentrations in summer followed by lower concentrations in spring and winter. Similarly, seasonal mean TN% in cattail and bulrush plant tissues was similar to that of TP in all the wetlands. Also, seasonal As concentrations (µg/g) in cattail at the LVW wetlands appeared slightly higher in summer followed by spring and winter seasons but were not statistically significant. Three of the presently studied wetland sites (LVW, HD and PW) were somewhat similar in that winter samples (not particularly fall) generally had higher Se concentrations in shoots for both plant types. This may be because of higher volatilization of Se in summer and spring season. These differences did not result in direct correlations with sediment and water data.

2.4.5 Ecosystem Function of Wetlands

Comparison of annual average nutrient storage in standing plants biomass showed that nutrient removal from the LVW wetlands was significantly higher than from the FW wetlands. This can perhaps be attributed to higher productivity (and thus more efficient nutrient removal) by cattails in the LVW wetlands. The LVW and FW wetland plants stored ~1357 kg/ha/yr of nitrogen and 257 kg/ha/yr of phosphorus, respectively. Also, the LVW wetland plants sequestered ~ 66 kg/ha/yr phosphorus compared to 15 kg/ha/yr at the FW plants (Table 2.2). However, based on the annual average nutrient storage in plants (kg/ha/yr) in the HD and PW wetlands, it was calculated that the HD and PW wetlands plants stored ~1612 and ~441 kg/ha/yr of nitrogen, respectively. Similarly, the HD wetland plants sequestered ~147 kg/ha/yr phosphorus compared to ~18 kg/ha/yr at the PW wetlands. Better ecosystem function of the HD and LVW wetlands is not only due to higher plant biomass and nutrient concentrations but also due to the larger surface area of the wetlands. Metal removal efficiency among the four wetland plants suggests that the annual average As uptake was higher at the LVW wetland plants (0.53kg/ha/yr) compared to FW (0.05kg/ha/yr). Similarly, LVW wetland plants stored (0.35kg/ha/yr) Se which was also higher than at FW plants (0.04kg/ha/yr). It appears that the larger the surface area of wetland vegetation, the higher the metal accumulation in plants and therefore higher the flux, suggesting that wetland acreage is equally important for better ecosystem function through pollutant removal. However, this was contradicted to some extent by the PW wetlands data which showed higher metal storage per unit area than any other wetlands in present study (Table 2.2).

Based on plant removal potential (kg/hac/yr) and annual load of nutrients and metals, total acerage of wetlands (hac) needed for effective removal under the regulatory threshold was estimated for four studied wetlands. Among the four wetlands, average annual TN conc. in LVW inlet water (14.7 mg/L) was measured above the regulatory provision for drinking water standard (10 mg/L) as defined by EPA. The wetlands acreage would need to be increased by four folds to meet the TN conc. requirement.

Similarly, to meet TP conc. under regulatory conc. both LVW and HD wetlands acerage would need to be increased by two folds. Annual average As conc. in PW wetland inlet water column (13.2 µg/L) was measured slightly above the regulatory threshold for aquatic life (10µg/L) as defined by EPA. The PW wetland would need to be at least 3.4 hac to reduce the As conc. under the regulatory standard. Annual average Se conc. in LVW wetland inlet water (3.2 µg/L), PW (10.9 µg/L), and FW (8.5 µg/L) were higher than the standard for aquatic life (2 µg/L). To reduce the Se conc. to the regulatory standard level, wetlands acerage would need to be increased to TN conc. level in all wetlands. The above estimates are done based only on TN, TP, As and Se in wetlands vegetation stored in kg/hac/yr therefore, do not consider other means of removal. Wetlands provide various other pathways for nutrients and metals removal such as volatilization, sedimentation and organic/inorganic transformation; therefore caution should be applied while interpreting these numbers.

Despite clear evidence that nutrients and metals are taken up by the plants in present study, it is puzzling that any significant water quality improvements are not seen between inlets and outlets. This could be because of the short residence time of water or short distance between inflow/outflow sampling locations which needs further investigation in future studies. However, it does appear that annual harvesting of the plants from these wetlands would provide significant removal of nutrients and metals.

Table 2.2 Inflow source, annual average nutrient and metal concentrations in water versus annual average nutrient and metal removal by plant (kg/ha/yr) at the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetlands at the City of Henderson Water Reclamation Facility (HD), and Pittman Wash Pilot Wetlands (PW). Digits after ± sign indicate standard errors.

Site	Nutrient Concentration of Water (mg/l)		Metal Concentration of Water (µg/l)		Nutrient Removal by Plant (kg/ha/yr)		Metal Removal by Plant (kg/ha/yr)	
	TN	TP	As	Se	TN	TP	As	Se
LVW	14.7±0.2 (n=48)	0.13±0.05 (n=30)	7.1±.4 (n=27)	3.2±0.1 (n=36)	1357	66	0.53	0.35
HD	6.5±0.9 (n=37)	1.41±0.10 (n=24)	3.4±0.1 (n=16)	1.9±0.1 (n=14)	1613	147	0.21	0.44
PW	9.02±0.1 (n=41)	0.005±0.01 (n=15)	13.2±0.5 (n=40)	10.6±0.1 (n=38)	441	18	0.41	0.71
FW	3.58±0.1 (n=65)	0.04±0.01 (n=22)	6.1±0.3 (n=28)	8.5±0.1 (n=12)	286	15	0.05	0.04

2.5 Conclusions

Constructed and naturally created wetlands in the Las Vegas Valley watershed were studied to understand their potential for pollutant removal. Significant removal of nutrients was found in the wetlands receiving high nutrient loads and both plant species in the four wetlands sites were quite efficient in taking up large amounts of nutrients and metals. The nutrient removal capacity of a wetland system was more dependent on individual plant biomass irrespective of plant type, i.e., on the size of individual plants or plant density. The nitrogen concentration was higher in above-ground plant parts but the phosphorus was higher in the below-ground parts, which suggests that harvest of the root system would be necessary for maximum phosphorus removal, but an above-ground harvest would be sufficient for nitrogen removal from four different wetlands systems. Plant nutrients in the four wetland sites correlated well with ambient nutrient concentrations in the sediments and water columns, irrespective of the type of plants present. Overall, this study suggests that different plant species have different capacities to take up nutrients, with these capacities mostly determined by the ambient nutrient and hydrologic conditions. Bulrush species seem particularly efficient for taking up metals such as As and Se, as compared to cattails. Also, the below-ground plants for both species seemed to store metals more efficiently than above-ground parts. Higher metal accumulation in the PW wetlands plants suggested that there is a potential for wildlife exposure. Better information on the bioaccumulative properties of the bulrush species found in the wetlands in this study might provide clues for Se removal using existing wetland plants in these wetlands. These findings have important implications for enriching ability to engineer ecological solutions to problems associated with nutrient-rich wastewater and to implement sustainable wetlands management plans.

CHAPTER 3

VEGETATION ASSESSMENT FOR NUTRIENT UPTAKE POTENTIAL BY MACROPHYTES IN SEMI-ARID WETLANDS, SOUTHERN NEVADA

This paper analyzes the structural and functional attributes of *Phragmites australis* (Cav.) Trin. ex Steud. and *Typha domingensis* Pers. for the best ecosystem services in arid and semi-arid wetlands. The entire LVW vegetation was analyzed through GIS mapping and ground truthing to estimate frequency and coverage of *P. australis* Vs. *T. domingensis*. The results from this study compared with the past data showed that *P. australis* population is increasing in areas with altered hydrology and high nutrient inputs.

In addition to its structural dominance, our data showed that *P. australis* play significant role for nutrient storage in large scale wetlands. The average above-ground biomass of *T. domingensis* varies from 5.6 to 11.1 kg dry weight (DW) m^{-2} and from 2.5 to 6.3 kg DW m^{-2} for *P.australis*. The net above-ground standing stock of nutrients in the Las Vegas Wash wetland was estimated to be approximately 26418.7 kg TN and 1264.1 kg TP for *P. australis* and to be approximately 5183.8 kg TN and 272.8 kg TP for *T. domingensis*. Despite management concerns over *P. australis* growth, they fared quite well in nutrient storage in LVW wetlands in comparison to *T. domingensis*. The substantial amount of nitrogen uptake by *T. domingensis* and *P. australis* in LVW wetlands suggests that both macrophytes can be utilized for water quality improvement. Finally, we compared the short-term functions and processes of macrophytes for nutrient removal potential among wetlands in semi-arid and humid regions.

3.1 Introduction

Riparian ecosystem function and structure have been changed dramatically over the past century in the Southwestern United States (Goodwin et al., 1997; Patten, 1998; Stromberg, 2001). During the past few decades, the modification of riparian ecosystems due to increased flow in urban streams of Desert Southwest is of widespread management concern (Goodwin et al., 1997). The increased flow resulting from urbanization coupled with extreme storm events creates extensive soil erosion, channel incision, nutrient enrichment, and threatens the existence of natural wetlands in riparian corridor (Bedford et al., 1999; LVWCC, 2010). Previous studies have shown that wetlands are useful for wastewater treatment, and high levels of nutrient retention and primary production (Engelhardt and Ritchie, 2001; Greenway, 2003; Vymazal, 2007; Kadlec and Wallace, 2009). However, there has been a concern over long term degradation of wetlands due to additional nutrient and hydraulic loadings from wastewater (Zhang et al., 2007).

Eutrophication is a common phenomenon caused by excessive nutrient loadings from anthropogenic sources, which also affects both structural and functional attributes of wetlands (U.S. EPA, 2002). The impact of nutrient enrichment on wetlands ecosystem services may depend on how wetlands vegetation influences biomass production and nutrient retention (Schlesinger, 1997; Bridgham et al., 1996; Pollock et al., 1998; Grace, 1999; U.S.EPA, 2002). Very little information exists on structural and functional attributes of aquatic vegetation in response to eutrophication and new management approaches in arid and semi-arid wetlands. The wetlands vegetation structure may shift as native species are outcompeted by species that take advantage of nutrient enrichment (Chambers et al., 1999; Galtowitsch et al., 1999; U.S. EPA, 2002). Emergent macrophytes *Phragmites australis* and *Typha* species frequently dominate the nutrient rich wetlands in most of the climatic regions (Urban et al. 1993; Mayerson et al., 2000). Since both species prefer a similar habitat, dominance of *P. australis* over *Typha* species is attributed to its efficient root development and the adaptability of its roots and rhizomes to fluctuating water tables (Clevering, 1999; Meyerson et al., 2000, Chun and Choi, 2009). Rapid expansion of invasive *P. australis* and *Typha* species into natural habitat and conversion of those habitats to monocultures pose a threat to wetlands globally, compelling researchers to investigate the mechanisms of invasion and dominance (Mayerson et al., 2000; Seabloom and van der Valk, 2003; Hager, 2004). Further, understanding these dynamics is critical to achieving and sustaining optimal treatment performance in constructed wetlands (Thullen et al., 2005).

Previous research has identified that the role of macrophytes in small-scale constructed wetlands could have a positive influence on nutrient removal from wastewater discharge (Reddy et al., 1989; Cooper et al., 1996; Brix, 1997; Templer et al., 1998; Meyerson et al., 1999; Findlay et al., 2002; Scholz, 2006; Kadlec and Wallace, 2009). The use of constructed wetlands for secondary wastewater treatment has been tested widely in recent years, especially to reduce nitrogen (N) and phosphorus (P) loads (Vymazal, 2007). However, little information is available on whether aquatic macrophytes could significantly contribute to biogeochemical cycling of large natural and artificially constructed wetlands. Plant uptake represents an important pathway for nutrient removal if periodic harvesting and removal of biomass is practiced (Paranychianakis et al., 2006). Nutrient removal can be optimized by improvements in plant selection and cultivation; however, differences in uptake performance may exist among different plant species. (Tanner, 1996; Mitsch and Gosselink, 2000; Vymazal, 2007; Maine et al., 2007; Brisson and Chazarenc, 2009). The nutrient removal performance of aquatic macrophytes such as *P. australis* and *Typha* species is maximized if above-ground biomass is harvested by the end of growing season (Asaeda et al., 2002; Toet et al., 2005). The understanding of vegetation function in nutrient uptake potential is particularly important to arid and semi-arid wetlands, because of the unique climatic and hydrological features (Mitsch and Gosselink, 2000). The aim of this study is to analyze the structural and functional attributes of dominant wetland vegetation for the best ecosystem services from large scale wetlands in arid and semi-arid regions. The entire LVW vegetation was analyzed through GIS mapping and ground truthing to estimate frequency and coverage of *P. australis* Vs. *T. domingensis*. Functional attributes for both species were analyzed for their nutrient storage potential. Finally, the results from this study were compared to nutrient uptake potential of aquatic macrophytes in humid regions. The findings from this study will help to better understand the vegetation function for treatment performance in arid and semi-arid wetlands.

3.2 Material and Methods

3.2.1 Study Area

Present research was conducted in the riparian corridor of downstream Las Vegas Wash, which receives wastewater discharge from the city of Las Vegas. The vegetation study was made in eight representative wetlands areas of which six are located in the main channel and were created after the construction of bend weir structures. The remaining two sites are separated from the main channel and being used as a wetlands park (Figure 3.1, Table 3.1). The representative sampling sites were established after analyzing an ARC Info database of wetlands in the Las Vegas Wash based on the satellite image obtained from USGS. A total of eight sampling sites were visited to adjust the plant coverate data in the field to accommodate altered wetland morphology and fixed the requisite sample size.

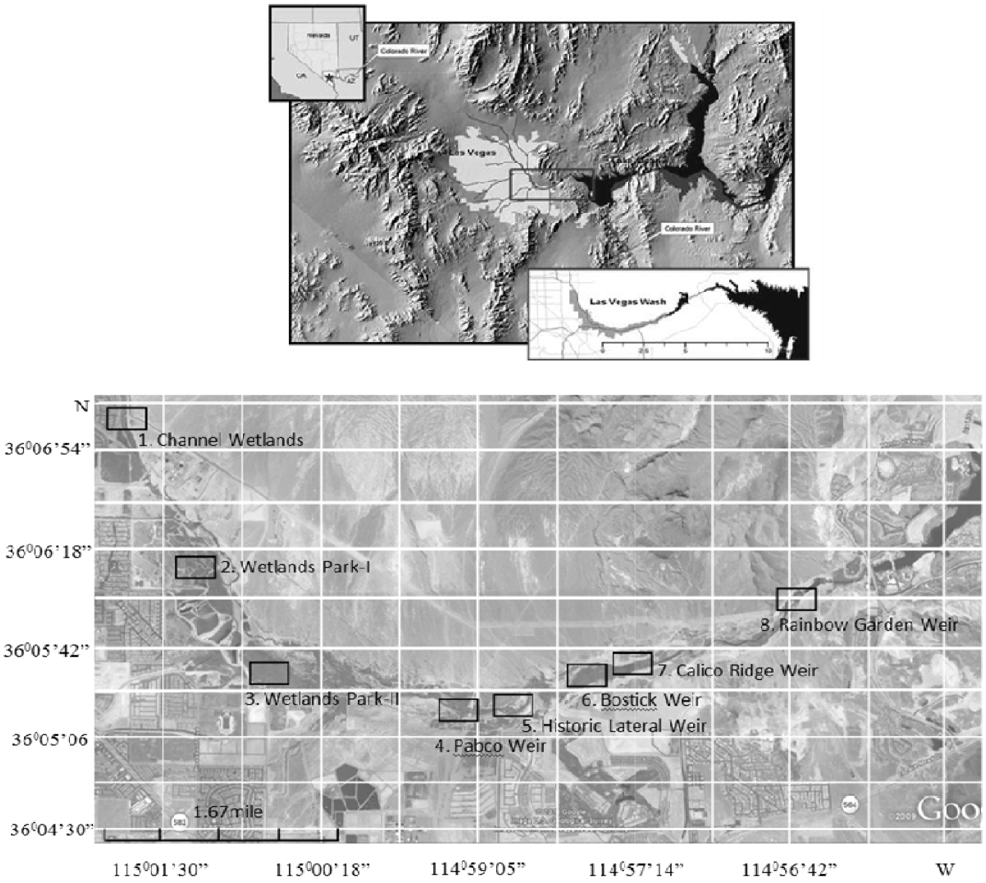

Figure 3.1 Map showing the eight vegetation sampling sites in the Las Vegas Wash

3.2.2 Vegetation Mapping

ArcInfo versions of ArcCatalog 9.2 and ArcMap 9.2 (ESRI, Redlands, CA) were used to analyze the vegetation data. A "Google Satellite Image 2009 provided by USGS" was transferred into a JMP file format. We defined a raster dataset using map coordinates, and assigned a coordinate system for georeferencing. Arc GIS editing tools were used to create polygon features and line features for vegetation classification from the spatial image data. The spatial image scale of 1:105844 was considered to create polygon feature. The physiognomy of different vegetation types which is characterized by a community's structure and form is the basis for creating polygons. The formation class uses structural attributes of the community including relative cover and height to separate vegetation into several categories. Because wetlands are protected by a variety of laws, regulations, and executive orders, it was important to identify institutional boundaries. The wetland determinations in this study were made according to the U.S. Army Corps of Engineers (Corps) wetland delineation manual (USCOE, 1987) and the procedure adapted from the vegetation study of Eckberg and Shanahan (2009).

Table 3.1 Wetlands sampling sites, area, and land use patterns in the Las Vegas Wash

Sampling site	Wetlands Area(ha)	Land Use
1.Channel Wetlands (CW)	2.53	In channel naturally created wetlands
2.Wetlands Park-I (WP-I)	15.87	Naturally created and constructed wetlands
3.Wetlands Park-II (WP-II)	18.30	Naturally created and constructed wetlands
4.Pabco Weir (PB Weir)	2.20	Naturally created wetlands along the upstream and downstream of bendweir structure
5.Historic Lateral Weir (HL Weir)	2.53	
6.Bostick Weir (BO)	4.81	
7.Calico Ridge Weir(CR Weir)	4.74	
8.Rainbow Garden Weir (RG Weir)	1.23	

3.2.3 Vegetation Sampling

Attributes including vegetation cover, frequency, and density among the eight wetland sites were estimated following the quadrant method (Barbour et al., 1999). All measurements were made during the month of July 2010. The line intercept method was used to determine the spatial distribution of the wetland vegetation. Plants species and genus were identified, depending on the apparent morphology in the field. All sampling sites were designed along transects nominally perpendicular to the water gradient. A total of 40 quadrants (0.5m ×0.5 m) were placed in a stratified random design to estimate vegetation frequency, cover, and density from each sampling site (Barbour et al., 1999). The “modified Broun Blanquet” cover class method was adopted using six cover categories: 1= <1%; 2 = 1-5%; 3 = 5-25%; 4 = 25-50%; 5 = 50-75%; 6 = 75-100% (Barbour et al., 1999). Five line intercepts 20 m in length were established at every 20 m interval in each wetland site for the vegetation distribution study.

3.2.4 Plant Biomass Analysis

The changes in wetland functions that occur in response to nutrient enrichment include increased plant biomass. The measurement of above-ground biomass in the present study corresponds with the maximum plant growth period (i.e. July 2009 and July 2010). All species from five random quadrants (0.5 m × 0.5 m) from each sampling site were harvested for a biomass study. Stems were counted in each quadrant, clipped to the ground, and separated into target and non-target species. Heights and wet mass of all stems from both categories were measured immediately following harvest. Individual stems were labeled and dried separately to determine the biomass (g m^{-2}). Plant biomass was calculated following methods described in APHA (2005) by storing samples for 72 hours at 70^{o}C or until a consistent dry weight was obtained. The resulting biomass is expressed as vegetation per unit area, also known as standing crop.

3.2.5 Plant Nutrient Analysis

For the nutrient analysis, above-ground and below-ground plant tissues from five random quadrants (0.5 m× 0.5 m) in were sampled from all eight sampling sites in July 2010. The random quadrants used for plant biomass estimation were also used for nutrient [total nitrogen (TN) and total phosphorus (TP)] analysis. Similarly, *T. domingensis* plant tissue was sampled in July 2009 following the same procedure. Dry plant samples were separated into roots, stems, and leaves prior to sub-sampling for nutrients. A Cyclone Sample Mill (UDY Corporation, Fort Collins, Colorado) was used to grind dry plant tissue to a homogeneous sample of approximately 1 mm in size for nutrient analyses. Plant tissue TP content was analyzed from 1 g dry subsamples using the colorimetric method (APHA, 2005). Similarly, plant tissue TN content was analyzed on a dry subsample (~1 g) using a PerkinElmer 2400 CHN analyzer.

3.2.6 Statistical Analysis

All data were tested for normality using Shapiro-Wilk test. One-way analysis of variance (ANOVA) was used to assess the effect of nutrient concentrations among different plant species. ANOVA-detected differences were compared using the Tukey pairwise comparison test. For all of the tests, p-values < 0.05 (95% confidence interval) were considered significant. Spatial distribution of wetland vegetation and species richness among sampling sites were examined using linear regression. Statistical analyses were carried out using JMP software (SAS Institute, Cary, North Carolina).

3.3 Results

3.3.1 Vegetation Mapping

The physiognomy of different vegetation types, community structure, and forms were described by vegetation mapping. Seven vegetation classes were identified from the Arc GIS database generated for the Las Vegas Wash. Polygon features of Arc Info were used to estimate the areas as well as vegetation cover for each vegetation class. Furthermore, the polygons representing wetland areas of herbaceous vegetation class were identified. A total of 34 polygons representing herbaceous vegetation from the Las Vegas Wash wetlands comprised approximately 0.89 km^2. The herbaceous classes were further separated into subclasses by persistence (annual or perennial) and growth form (graminoid, forb, or hydromorphic). Herbaceous wetland vegetation having similar physiognomic features and which were found along particular hydrologic conditions and topographic positions were defined. They were dominated by four emergent macrophytes, namely *P. australis, Typha domingensis, Schoenepletus spp,* and *Distichlis spicata*, which shared semi-permanent and permanently flooded habitats.

3.3.2 Vegetation Distribution

The vegetation cover study showed the dominance of *P. australis* on six out of eight wetland sites in the Las Vegas Wash (WP-I: 46.5%, WP-II: 72.2%, PB Weir: 42.7%, BO Weir: 59.7%, HL Weir: 36.3%, and CR Weir: 30.2%, Figure 3.2a) . However, *T. domingensis* had the highest plant cover measured for two wetland sites (CW: 33.1%, RG Weir: 39.2%), remained co-dominant on other wetland sites (Figure 3.2a). The frequency distribution for both species showed a trend similar to that of vegetation cover. The frequency distribution of *P. australis* was measured highest on five wetland sites (WP II: 82.1%, PB Weir: 66.6%, BO Weir: 74.2 %, HL Weir: 64.1 %, and CR Weir: 58.3%), followed by *T. domingensis* in two wetland sites (CW: 54.2 %, RG Weir: 56.7%), (Figure 3.2b). The highest frequency of aquatic weeds (48.3%) was measured at WP I wetland sites; however, the small size of those species resulted in less cover percentage (Figure 3.2b).

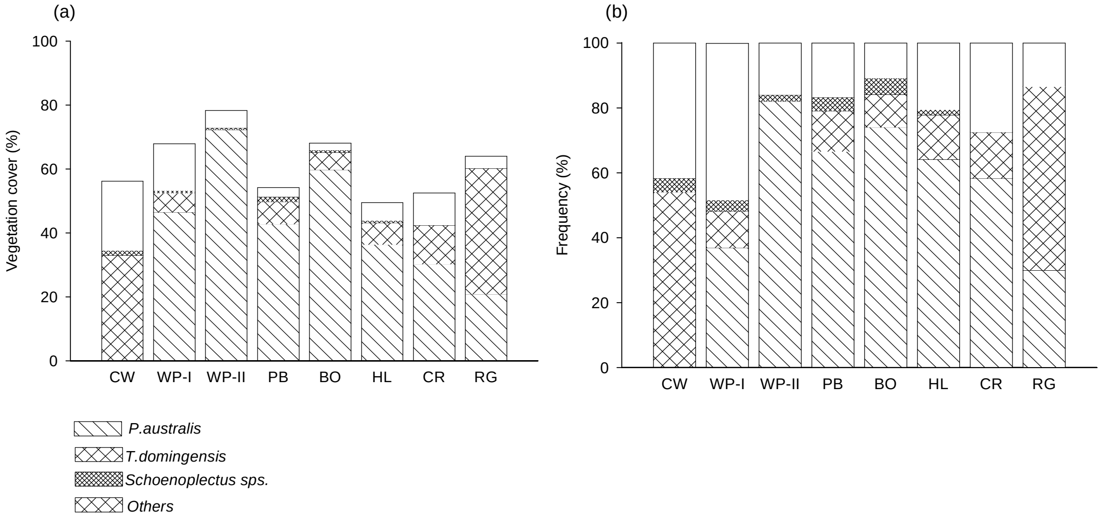

Figure 3.2 Estimates of: a) vegetation cover (%) using cover class method (Modified Broun Blanquet method); and, b) frequency (%) among various sampling locations in the Las Vegas Wash wetlands.

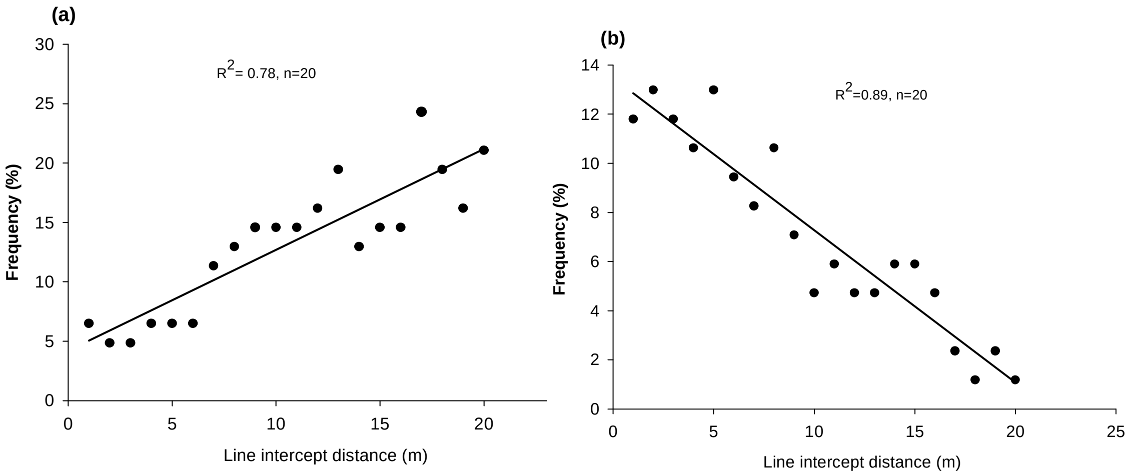

Figure 3.3 Linear regression between the line intercept distance (m) and frequency (%) of a) *P. australis* and b) *T. domingensis* measured form the centre of the water gradient perpendicular toward the bank areas of wetland sites in the Las Vegas Wash wetlands.

The frequency of *P. australis* and *T. domingensis* on the eight wetland sites was measured using a 20 m long line intercept. The species distribution pattern followed the habitat preferential trend among all studied sites. *P. ausralis* distribution was dominant along the stream bank areas, and decreased toward the center of the stream (Figure 3.3a). The frequency of *T. domingensis* species occurrence was high near the water and reduced significantly toward the bank areas (Figure 3.3b).

Average stem density for *T. domingensis* (approximately 40 m ^{2}) was significantly less than the stem density of *P. australis* (approximately 115 m^{-2}) on the eight wetland sites in the Las Vegas Wash. The stem density of *P. australis* was higher on the newly established stands and was lower on the wetlands sites with a relatively old stands. Stem density was also correlated to species richness. A negative correlation was observed between the *P. australis* stem density and species richness wetland sites in the Las Vegas Wash (Figure 3.4).

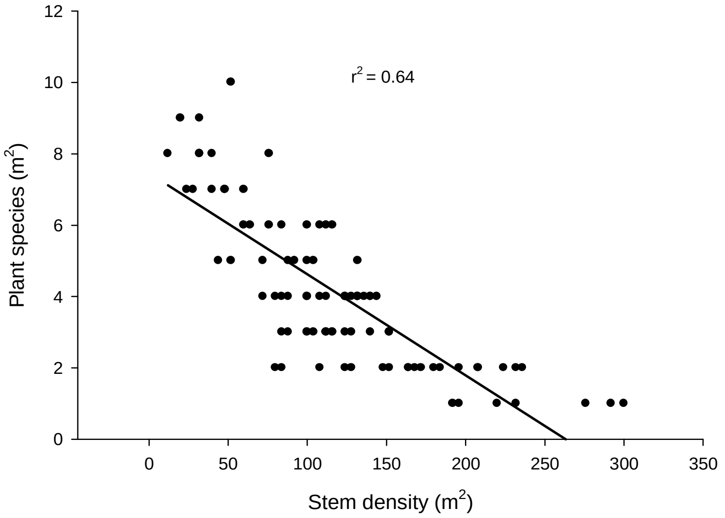

Figure 3.4 Linear regression between species richness and stem density of *P. australis* from the sampling sites in the Las Vegas Wash wetlands.

3.3.3 Plant Biomass

The highest amounts of above-ground biomass of *P. australis* throughout the Las Vegas Wash study area were recorded at BO Weir (6.1 ± 0.8 kg m^{-2}) and PB Weir (6.3 ± 0.8 kg m^{-2}) (Table 3.2). Similarly, the above-ground biomass for *T. domingensis* was highest at BO Weir (11.1 ± 1.2 kg m^{-2}). Average above-ground biomass values ranged from 5.6-11.1 kg m^{-2} for *T. domingensis* to 2.8-6.3 kg m^{-2} for *P. australis* throughout the eight sampling sites (Table 3.2). Total above-ground biomass storage potential was approximately 874 tons for *P. australis* and approximately 341 tons for *T. domingensis* from all wetland sites in Las Vegas Wash (Table 3.2). Among all studied sites, the above-ground biomass from *T. domingensis* was significantly higher than *P. australis*. In the wetland sites examined in this study, the harvest of *P. australis* above-ground biomass could result in a removal of 520-1327 kg TN ha^{-1} and 25-61 kg TP ha^{-1}. Similarly, this study showed that *T. domingensis* could store a total of 852- 1682kg TN ha^{-1} and 45-88 kg TP ha^{-1} during their (its) maximum growing period.

Table 3.2 Eight sampling locations showing above-ground biomass (kg m^{-2}) for *T.domingensis* and *P. australis* measured in the Las Vegas Wash wetlands areas.

Site	Wetland Area	*Typha* AG-Biomass		*Phragmites* AG-Biomass	
	(m^2)	(kg m^{-2})	(Ton)	(kg m^{-2})	(Ton)
HL Weir	25292.8	7.6±1.1	12.9±1.8	2.8±0.4	17.4±4.9
CR Weir	12302.4	7.7±1.1	9.8±1.3	2.5±0.3	6.6±1.8
RG Weir	47388.7	6.7±0.3	125.5±5.9	3.9±0.5	27.9±7.5
BO Weir	48157.6	11.1±1.2	28.7±3.2	6.1±0.8	120.8±32.1
PB Weir	22055.4	5.6±0.4	8.7±0.6	6.3±0.8	40.9±10.2
WP-I	158758.3	7.4±1.1	70.3±9.9	2.9±0.4	145.3±40.1
WP-II	253211.8	0	0	4.1±0.5	515.8±143.3
CW	33305.6	7.8±0.4	85.2±4.3	0	0
Total (Ton)			341.2		874.4

3.3.4 Plant Nutrient

Above-ground plant tissue concentration for *P. australis* for TN (20.9 mg g^{-1}) and TP (1.0 mg g^{-1}) was higher than *T. domingensis* for TN (15.2 mg g^{-1}) and TP (0.8mg g^{-1}) (Table 3). For below-ground tissues, *P. australis* tissue measured higher in concentration of TN (12.3 mg g^{-1}) than *T. domingensis* (11.4 mg g^{-1}). However, the below-ground tissue TP concentration of *P. australis* (0.8 mg g^{-1}) was less than TP concentrations measured in *T. domingensis* (2.2 mg g^{-1}) (Figure 3.5 a, b).

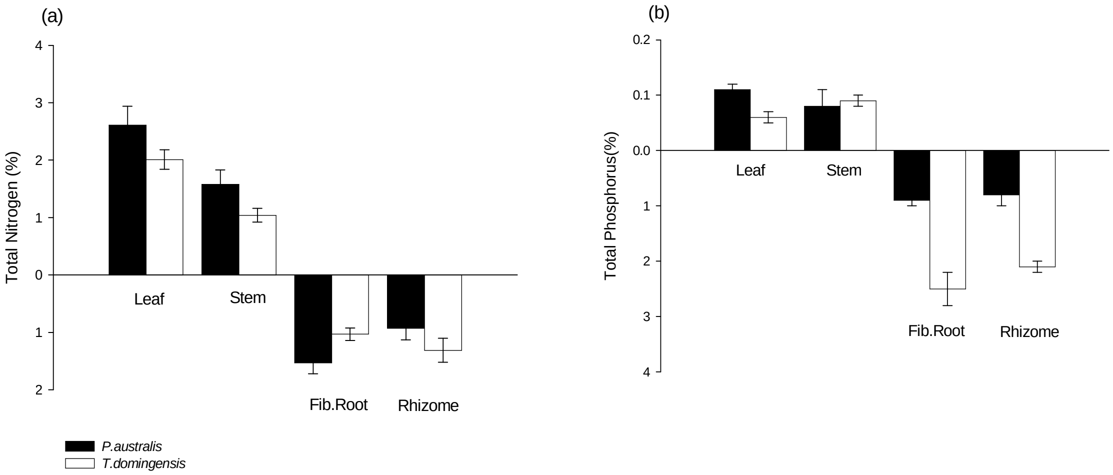

Figure 3.5 Average a) total nitrogen uptake (g m^{-2}) and b) total phosphorus uptake by individual plants (*P. australis* and *T. domingensis*) from sampling locations in the Las Vegas Wash wetlands. Error bars represent standard errors.

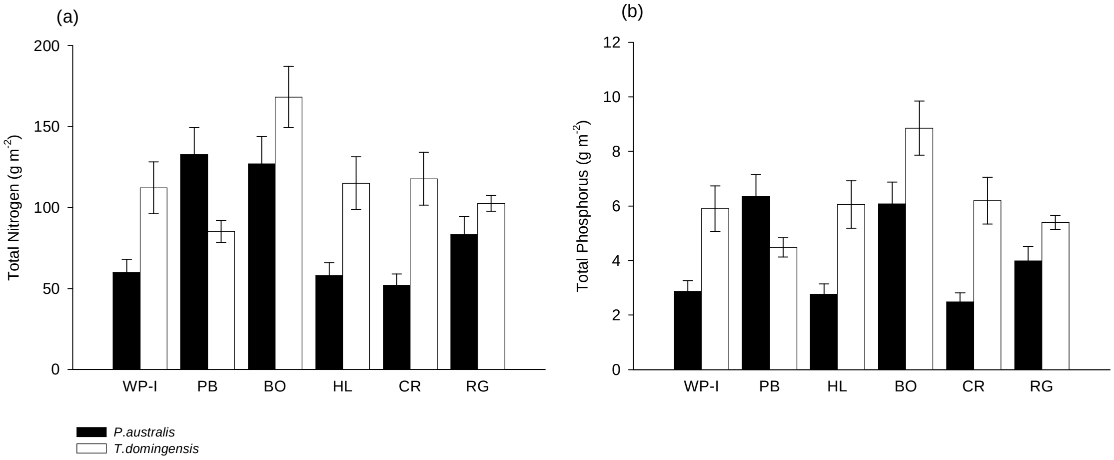

Figure 3.6: Average a) total nitrogen uptake (g m^{-2}) and b) total phosphorus uptake by individual plants (*P. australis* and *T. domingensis*) from sampling locations in the Las Vegas Wash wetlands. Error bars represent standard errors.

Above-ground leaf tissue of *P. australis* showed higher TN concentrations than *Typha* species. The average TN and TP uptake (g m^{-2}) measured significantly higher in *T. domingensis* than *P. australis* for among most of the sampling sites (Fig 3.6a, 3.6b).

Average TN and TP above-ground nutrients uptake potential for *T. domingensis* among different studied sites in the Las Vegas Wash varied from 85.2 - 168.2 TN g m^{-2} and 4.5 - 8.8 TP g m^{-2}. The corresponding results in *P. australis* were 52.1-132.7 TN g m^{-2} and 2.2 - 6.8 TP g m^{-2}.

Table 3.3 Eight sampling locations showing total area (m^2), and above-ground storage (kg) potential for *T. domingensis* and *P. australis* in the Las Vegas Wash wetlands areas.

Site	*Typha* AG storage		Total Area	*Phragmites* AG storage	
	TN (kg)	TP (kg)	(m^2)	TN (kg)	TP (kg)
HL Weir	196±28.0	10.3±1.5	9200.3	533.6±73.1	25.4±3.4
CR Weir	148±20.1	7.8±1.1	3875.3	201.6±26.7	9.6±1.2
RG Weir	1908±90.4	100.4±4.7	9892.4	824.9±109.5	39.4±5.2
BO Weir	435±48.1	22.9±2.5	28774.2	3556.3±481.1	174.9±23.1
PB Weir	131±10.1	6.9±0.5	9428.7	1251.3±157.6	59.8±7.5
WP-I	1068±152.2	56.2±8.1	73822.5	4428.1±601.7	211.8±28.7
WP-II	0	0	182945.5	15524.6±2102.9	742.7±100.6
CW	1294±65.1	68.1±3.4	0	0	0
Total(kg)	5183.8	272.9		26418.7	1264.1

Nutrient uptake for both species was proportionate with plant biomass rather than tissue nutrient concentration. Among the eight wetland sites, above-ground biomass (kg m^{-2}) for *T. domingensis* was higher than *P. australis* except at PB Weir. However, the total nutrient (TN and TP) uptake potential in most of the wetlands sites was dominated by *P. australis,* due to the large surface area covered by this particular species (Table 3.3). The net above-ground standing stock of nutrients in the Las Vegas Wash wetlands was approximately 26418.7 kg TN and 1264.1 kg TP for *P. australis* and approximately

5183.8 kg TN and 272.8 kg TP for *T. domingensis,* if harvested during the peak growing period.

3.3.5 Nutrient dynamics in humid and semi-arid wetlands

To provide a context for these wetland vegetative results, comparative values from other wetlands were obtained, and the nutrient distribution functions of common aquatic macrophytes being used for wastewater treatment purposes from several climatic regions were compared to those recorded in this study. Humid region wetlands have variable functions within temperate and tropical climates, but structural and functional attributes in those environments are more frequently studied than arid and semi-arid wetlands. Standing biomass and nutrient stock values measured in Las Vegas Wash wetlands were comparable to values for constructed wetlands being used for secondary and tertiary wastewater treatment in humid regions. The above-ground biomass from humid (tropical and temperate) and semi-arid regions including Mediterranean climates ranged from 1320 g m^{-2} to 4046 g m^{-2} for *Phragmites* species, and from 1045 g m^{-2} to 4003 g m^{-2} for *Typha* species (Behrends et al., 1994; Ennabili et al., 1998; Greenway, 2002; Obarska -Pempkowiak and Ozimek ,2003; Ciria et al., 2005; Fernandez and de Miguel, 2005; Toet et al., 2005; Vymazal and Kropfelova, 2008; Alvarez-Cobelas and Cirujano, 2007; Lesage et al., 2007; Maine et al., 2007; Maddison et al., 2009) (Figure 3.7). The above-ground biomass values for both *Typha* and *Phragmites* stands measured in the Las Vegas Wash during the maximum growth period were higher than comparable values reported for humid and Mediterranean region wetlands. Net primary productivity (NPP) for *Phragmites* species in temperate regions (mostly in Europe) were less than NPP values recorded in semi-arid and tropical region wetlands (Maddison et al., 2009).

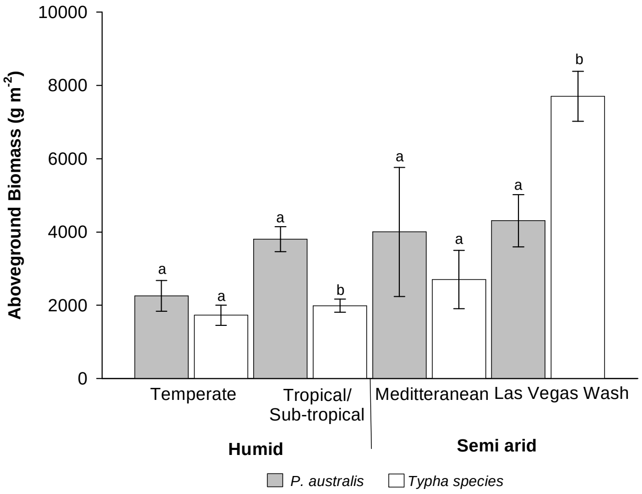

Figure 3.7 Maximum above-ground standing biomass for *Phragmites* and *Typha* species in wetlands constructed for wastewater treatment in humid and semiarid regions. Letters above bar denote significant difference based on pairwise (Tukey HSD) comparisons. Error bars represents standard errors.

In general, the N and P tissue concentrations showed less variation among humid and arid region wetlands. The above-ground tissue TN concentration in the humid tropical climates (1.5-3.9%) and temperate climates (0.8-3.9%) were slightly higher than in semi-arid climates (1.5-2.1%), but were similar to natural wetlands in the humid regions (Figure 3.8a). The above-ground plant tissue TN and TP concentration showed a similar trend among different wetlands in humid and semi-arid regions (Kadlac and Knight, 1996; Vymazal, 1999; Johnston, 1991) (Figure 3.8b).

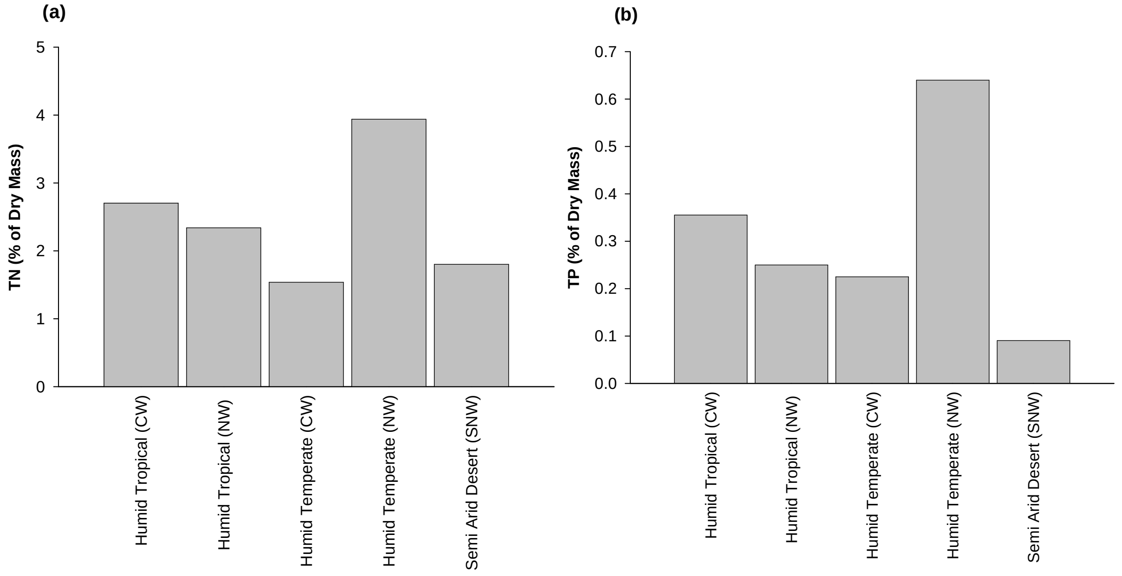

Figure 3.8 Dry mass nutrient concentrations for a) total nitrogen and b) total phosphorus in above-ground tissue of wetland plants from constructed wetlands (CW), natural wetlands (NW), and semi natural wetlands (SNW) of various climatic regions.

Both *Phragmites* and *Typha* species were estimated to have a similar amount of nutrient stock among different climatic regions. The TN standing stock reported for humid temperate regions (32.2-250 g TN m^{-2}) was higher than for humid tropical (15.8-156 g TN m^{-2}) and semi-arid (52-132 g TN m^{-2}) regions for *Phragmites* species (Figure 3.9a). The range of TN standing stock values for *Typha* species in humid tropical regions (71-250 g TN m^{-2}) were higher than for humid temperate (32-120 g TN m^{-2}) and semi-arid region wetlands (85-168 g TN m^{-2}) (Reddy and De Busk, 1987; Brix and Schierup, 1989; IWA, 2000; Greenway and Wolley, 2001; Maddison et al., 2009). The P standing stock for *Typha* and *Phragmites* species reported for humid tropical and humid temperate wetlands were higher than those in semi-arid region wetlands (Figure 3.9b) (Tanner, 1996; Muelman, 2002; Toet et al., 2005; Vymazal and Kropfelova, 2008).

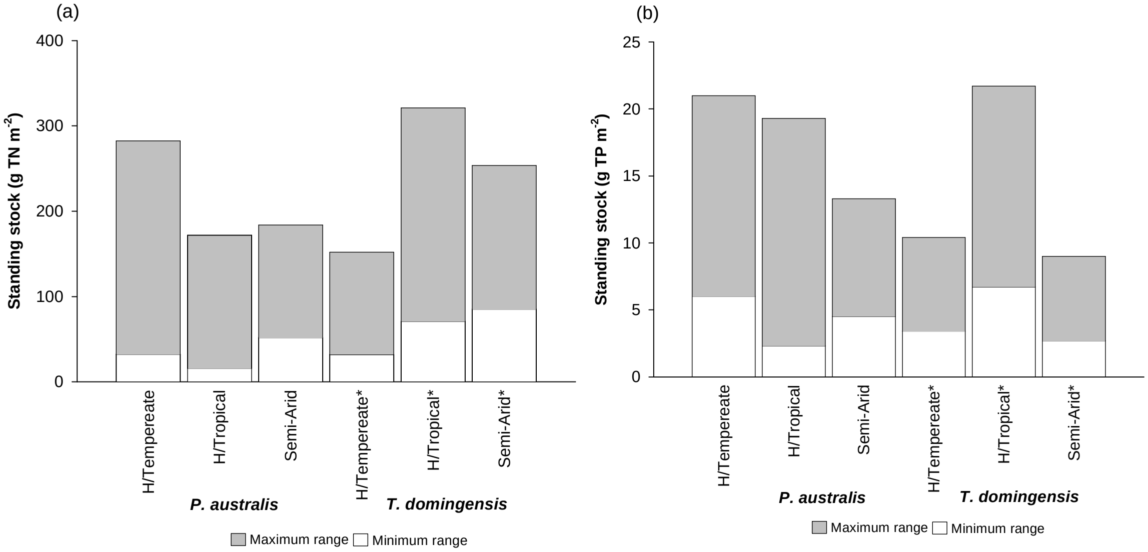

Figure 3.9 Ranges of standing stock for a) total nitrogen (g TN m^{-2}) and b) total phosphorus (g TP m^{-2}) for *Typha* and *Phragmites* species in natural wetlands and constructed wetlands for wastewater treatment in different climatic regions.

The annual average inflow volume at the Las Vegas Wash (for the period 2009 to 2010) was 8.5 cubic meter per second and annual mean wastewater concentration for N and P was 14.2 ± 0.2 mg L^{-1} and 0.13 ± .003 mg L^{-1} ,respectively (SNWA, 2010). In this study, the Las Vegas Wash wetlands averaged 102 ± 10 g TN m^{-2} and 5.12 ± 0.5 g TP m^{-2} standing stock during the maximum growth period (from June 2009 to July 2010). The average nutrient load in Las Vegas Wash was significantly higher than comparable values reported for typical constructed wetlands designed for wastewater treatment. The applied nutrients load in the Las Vegas Wash wastewater was estimated to be 11244 g TN m^{-2} yr^{-1} and 176.5 g TP m^{-2} yr^{-1}, respectively.

3.4 Discussion

3.4.1 Vegetation Mapping

Mixed marsh vegetation, dominated by the emergent species e.g. *T. domingensis,* was the major component of wetlands in the Las Vegas Wash, covering 2.14 km^2 in 1975. Consecutive flood events in 1983 and 1984 created a defined channel in the wash system and eroded the existing marsh vegetation. In 1982, *T. domingensis* covered 1.22 km^2 and 0.53 km^2 by 1986 (USBR, 1987). *T. domingensis* cover further reduced to 0.26 km^2 in 1995 and to only 0.06 km^2 in 1998 (SNWA, 1999). Increased wastewater flow in the Las Vegas Wash led to excessive scouring and draining resulting in the rapid degradation of wetland habitat. Nutrient enrichment, on other hand, had impaired the survival of native wetland species. The plant cover was only 0.21 km^2 for *P. australis* in 1975, extended to 0.65 km^2 in 1985, and further extended to 1.17 km^2 in 1995 (USBR, 1987). Such an increase of non-native *P. australis* is likely the result of nutrient enrichment and cultural eutrophication, as the Las Vegas Wash wetlands were initially dominated by native species. Management actions to control the *P. australis* invasion limited its extent to 0.64 km^2 by 1998 (SNWA, 1999). Native *T. domingensis* cover increased to 0.15 km^2 by 2005 after the construction of bend weirs for soil erosion control. The construction of bend weirs has stabilized the channel hydrology, provided extra space for *P. australis* extension, and the cover increased to 0.76 km^2 by 2005 (SNWA, 2010).

The present study shows the vegetation cover in Las Vegas Wash wetlands is dominated by *P. australis* and followed by *T. domingensis* as a co-dominant species. The historical trend and recent vegetation mapping of the Las Vegas Wash wetlands suggest the shifting trend in wetland habitat, which is simultaneously replaced by species that take advantage of altered hydrology and high nutrient input. The dominance of these species is often regarded as an indicator of nutrient enrichment and eutrophication (Davis, 1991; Craft, 1997; Chambers et al., 1999; Galatowitsch et al., 1999; US EPA, 2002).

3.4.2 Vegetation Distribution

Nutrient enrichment affects the vegetation structure through shifts in species composition, cover, and frequency. Both species (*P. australis* and *T. domingensis*) expanded to fill wetland areas in Las Vegas Wash made available after the construction of bend weir structures. These species take advantage of the higher nutrient input in Las Vegas Wash, while the hydrological attributes have limited the extent of *T. domingensis* compared to *P. australis*. In the eight wetland sites studied, the distribution of vegetation cover and frequency was compatible with the existing water gradients. The higher frequency and cover of *P.australis* at WP-I and WP-II was associated with low water flow volume and large surface areas. The higher frequency and cover of *T. domingensis* at RG Weir may have been the result of the well-designed bend weir, which created an extended backwater pool behind the structure.

The expansion of *P. australis* usually occurs on the upper fringes or elevated areas of wetland basins where the water tables are likely low (Marks et al., 1994; Hudon, 2004). The deeper roots and rhizomes of *P. australis* appear to be more adapted to both low and high water sites. In general, *P. australis* favors less waterlogged soil and low water table locations for sprouting. Since the rhizome has adapted to fluctuating water levels, *P. australis* shoot growth may first establish itself on sites with a low water table and expand to higher water levels through vegetative expansion of rhizomes (Amsberry et al., 2000; White and Ganf, 2002; Chun and Choi, 2009). The colonization of *P. australis* usually begins in raised mounds and then expands to *T. domingensis*-dominated depressions. In contrast, *Typha* species normally occurs in depressions where the water table is relatively high (Choi and Bury, 2003). *T. domingensis* has a shallower rooting depth and its potential for shoot production and growth requires a larger amount of biomass than does *P. australis*. This may suggest that the former can grow on sites with either low or high water tables, whereas the latter prefers an elevated table. Therefore, the expansion of *T. domingensis* in Las Vegas Wash is less likely since the variable water level from high to low is common in most of the wetlands location.

The dominance of the *P. australis* was also evident in the species richness values from wetland sites in the Las Vegas Wash. The species richness gradually decreased with the increase of *P. australis* stem density. The wetland sites, including BO weir and PB weir, exhibited less species diversity, which may be due to the monogeneric succession of *P. australis*. Several other studies have reported a decline in plant species richness with progressive enrichment in nutrients, as well as the dominance of aggressive species (Bridgham, 1996; Gustafsong and Wang, 2002; Wilcox et al., 2003; Frieswyk et al., 2007; Trebitz and Taylor, 2007).

3.4.3 Plant Biomass

Functional attributes of wetland vegetation were analyzed through estimation of TN and TP removal efficiencies for *P. australis* and *T. domingensis*, since both species accounted for more than 80% of total macrophytes cover in Las Vegas Wash wetlands. Above-ground biomass of *P. australis* stands (2.5-6.3 kg m^{-2}) in the present study were at the higher end of biomass values, ranging from 0.6-4.9 kg m^{-2}, observed from nutrient enriched natural and constructed wetlands reported in other studies (Toet et al., 2005; Meuleman et al., 2002; and Hosoi et al., 1998). The average above-ground standing stock of *P. australis* estimated in our study was also higher than that reported by Lesage et al. (2007) (1.5 kg m^{-2}) and Maddison et al. (2009) (0.6-1.3 kg m^{-2}) for treatment wetlands. Maximum above-ground biomass measured from *T. domingensis* (5.6-11.1 kg m^{-2}) in our study was at the higher end of stock values of 2.1 kg m^{-2}, 2.2kg m^{-2}, and 0.4-1.7 kg m^{-2} from semi-natural and constructed wetlands as measured by Toet et al. (2005), Fernandez and de Miguel (2005) and Maddison et al. (2009) of respectively.

High removal rates are possible when the vegetation is harvested at peak nutrient storage levels (Meuleman et al., 2002). However, several authors have suggested that harvesting *P. australis* in the early summer may negatively affect the long term vitality of the *P. australis* stand. The timing of the above-ground biomass harvest also affects annual rhizome resource allocation in the *P. australis* (Asaeda et al., 2005). According to Weisner and Granéli (1989) and Granéli et al. (1992), translocation of non-structural carbon is completed in the months of July-August. Thus, under the eutrophic conditions prevailing at the wetlands used for wastewater treatment, harvesting in September-October likely will not affect the long term vitality of the *P. australis* stand (Meuleman et al., 2002). Vegetation management will be more sustainable when above-ground standing stock in the Las Vegas Wash wetlands is harvested in the early fall. The net standing stock of *P. australis* was estimated at 874.3 tons, and 341.2 tons for T. *domingensis* in the

Las Vegas Wash wetlands. The higher net biomass of *P. australis* is the result of the large extent of the vegetation despite considerably a greater the biomass yield of *T. domingensis* stands. Moreover, a relatively high above-ground biomass of *P. australis* was measured at the two weir sites, BO weir and PB Weir, than among different wetland sites in the Las Vegas Wash. Such higher biomass is associated with higher density and the longer shoots attained by monogeneric stands in historically older wetland sites.

3.4.4 Plant Nutrients

In our study, above-ground tissue of *P. australis* had higher TN and TP concentrations storage than did the below-ground tissue. High N uptake might be the result of maximum nutrient translocation among the above-ground tissues during the peak growing season. The distribution of TN concentration was similar in *T. domingensis*, but higher TP concentrations were measured in below-ground tissue for the same period. The high TP concentration in below-ground tissue might be the result of late inflorescence in *T. domingensis* and delayed translocation. Previous research also reported that the higher P translocation and retranslocation during pre- and post-inflorescence was more efficient in *P. australis* than *T. domingensis* (Kühl et al., 1997, Meuleman et al., 2002, Toet et al., 2005). The nutrient removal capacity measured in the Las Vegas Wash wetlands was similar to capacities reported by IWA (2000), where they found the optimal removal of 200-2500 kg TN ha^{-1} and 30-150 kg TP ha^{-1} by emergent macrophytes in constructed wetlands. All Las Vegas Wash sampling sites were observed for alteration of vegetation structure and function asymptotically with increasing TN rather than TP. Our findings are consistent with results from the freshwater wetlands in the Midwestern United States where above-ground biomass and dominance of *T. domingensis* and *P. australis* species are also positively correlated with N enrichment (Craft et al., 2007).

Nutrient removal by shoot harvest in the Las Vegas Wash wetlands would be greatest when the maximum TN and TP standing stocks are reached during the peak growth months. However, shoot harvest during peak growth months will have detrimental effects on sustaining the vitality of the vegetation in the long run, since a considerable part of the transport process of non- structural carbohydrates and nutrients from the shoots to below-ground plant parts has not yet occurred. However, the harvest of the *P. australis* during the peak growing season could help to restrain its further dominance and create suitable habitat for *T. domingensis*. The wetlands in the Las Vegas Wash in absence of plant harvest might function as a nutrient sink, and high nutrient retention in the system would further promote eutrophication.

3.4.5 Nutrient dynamics in humid and semi-arid wetlands

Despite low plant tissue concentration and standing stock nutrients in semi-arid wetlands, the above-ground biomass in the Las Vegas Wash wetlands was higher than values reported for constructed wetlands in humid region. Most of the wetland sites in the Las Vegas Wash were created naturally and not specifically designed for maximum nutrient retention. Macrophyte turnover rate in humid tropical and semi-arid regions can be as high as 4-5 times per year due to the fact that warm climates favor a longer growing season. Turnover, in humid temperate environments, is usually one or two years which might be a reason for low amounts of standing biomass in temperate region wetlands

(Mitsch and Gosselink, 2000). Nutrient removal capacity of any constructed or natural wetland is the function of the nutrient stock with respect to the applied nutrient load. Due to the high disparity between standing stock and applied nutrient load, the nutrient removal by the Las Vegas Wash wetlands was limited to 0.90% and 2.9% of the total applied TN and TP load, respectively. However, the nutrient removal by harvesting above-ground standing biomass of common wetland plants (cattail, bulrush, common reed) among surface wetlands and constructed wetlands was high (4.3-21%) (Adcock et al., 1994; Hurry and Bellinger, 1990; Vymazal et al., 1999; Mueleman et al., 2002).

Humid and semi-arid region wetlands were not significantly different in their short-term structural and functional attributes. High variability exists for standing biomass, nutrient concentration, and standing stock within humid and semi-arid region wetlands. The performance efficiency of wetlands macrophytes in both arid and humid region depends on common variables including the quality and quantity of the wastewater input, hydraulic retention time, and biological and chemical processes. The short-term evaluation of wetland functions in both semi-arid and humid regions were adequate for nutrient removal in both constructed and naturally created wetlands (Mitsch and Gosselink, 2000). However, their function and processes over the long-term is particularly important in arid and semi-arid areas, where surface water levels fluctuate both seasonally and inter annually (Sanchez-Carrillo and Alvarez-Cobelas, 2000). Emergent macrophytes are well adapted to hydrological fluctuations and probably the best indicators of those dynamics. The rate of biomass decomposition in arid and semi-arid wetlands is higher than in humid wetlands, which subsequently increases the nutrients retention to the ecosystem and enhances the eutrophication process (Gumbricht et al., 2002).

3.5 Conclusion

Vegetation mapping in the Las Vegas Wash suggests the shifting of wetland habitats now dominated by *P. australis*, which take advantage of nutrient enrichment and altered hydrology. The structural attributes of vegetation, including plant cover and frequency, were consistent with the existing water gradients along the Las Vegas Wash. The expansion of *P. australis* habitat has readily surpassed *T. domingensis* habitat, it has established monogeneric stands, and its presence is associated with decreasing species richness. The change in structural attributes of vegetation, in turn, affects the ecosystem function by altering the nutrient biogeochemical cycling.

The above-ground biomass (kg m^{-2}) of *T. domingensis* was higher than *P. australis* in our studied wetland system. Plant tissue TN and TP concentrations were higher in *P. australis* above-ground plant tissue than in *T. domingensis*, but a greater biomass accumulation potential of *T. domingensis* resulted in higher nutrient storage per unit area. The net above-ground standing stock of nutrients in the Las Vegas Wash wetlands was estimated for *P. australis* (approximately 26418.7 kg TN and 1264.1 kg TP) and for *T. domingensis* approximately 5183.8 kg TN and 272.8 kg TP). The study concludes that in LVW, both *T. domingensis* and *P. australis* could be utilized for water quality improvement. It should be noted, however, plant uptake alone is not enough to improve water quality below regulatory thresholds from large scale wetlands, and managing dominant vegetation may be required for better nutrient removal efficiency.

The present study reviewed the short-term wetland functions among different climatic regions in order to provide a context for values obtained from the semi-arid Las Vegas Wash wetlands. Wetland function for biomass accumulation, nutrient concentration, and standing stock was not significantly different among humid and semi-arid region wetlands.

CHAPTER 4

ESTIMATION OF ROOT COHESION FOR DESERT SHRUB SPECIES IN THE RIPARIAN ECOSYSTEM OF ARID AND SEMI-ARID REGIONS AND ITS POTENTIAL FOR STREAMBANK STABILIZATION

Quantifying mechanical properties of native vegetation for streambank stability has remained a critical need of the Lower Colorado riparian revegetation effort. In the present study root cohesion is estimated for four representative native desert shrub species : *Artiplex lentiformis* (Torr.) S.Watson, *Lycium andersonii* A. Gray, *Larrea tridentata* (DC.) Coville, and *Allenrolfea occidentalis* (S.Watson) Kuntze to understand their suitability in streambank stabilization in the framework of a revegetation campaign. Field experiments were conducted to measure root length, root length density, root area ratio, and root tensile strength. Finally, the root cohesion values were assessed using a simple perpendicular model.

The root tensile strength (T_r) was greatest for *L. tridentata* (62.23 MPa) followed by *L. andersonii* (53.53 MPa), *A. lentiformis* (49.17 MPa), and *A. occidentalis* (35.03 MPa). The maximum root cohesion in the present study was estimated for *A. lentiformis* (97.6 kPa) followed by *L. andersonii* (89.3 kPa), *L. tridentata* (35.6 kPa), and *A. occidentalis* (34.8 kPa). Root cohesion values were also estimated using Fiber bundle model (FBM) and compared to the perpendicular root model of Wu et al. (1979). The comparative root cohesion values for root diameter (> 0.5 mm) suggest that Wu's model estimates are greater than those of the FBM model by a reduction factor ranges between 0.35 and 0.56 for presently studied species.

4.1 Introduction

The riparian areas of the Lower Colorado River Basin (LCRB) have undergone substantial degradation over the past century (Goodwin et al., 1997). Restoration activities have been increasing since 1990 to counteract the detrimental effects of flow regulation and altered hydrology (Follstad Shah, 2007). Channel narrowing and incision have been a severe issue in the major tributaries of the LCRB. These changes have been attributed to a number of factors including climatic and anthropogenic changes in the flow regime and invasion of exotic riparian species (Pollen-Bankhead et al., 2009). In a few of the important tributaries in LCRB including the Las Vegas Wash and the Lower Salt River, the increased flow discharge resulting from urbanization, coupled with extreme storm events has resulted in streambank retreat, extensive soil erosion, and downstream sediment deposition (LVWCC, 2010).

Federal, state, and local agencies have worked together since 2005 to implement a 50 year Lower Colorado River Multi-Species Conservation Plan to protect and maintain wildlife habitat along the LCRB in Nevada, Arizona, and Southern California. This plan includes the goal of creating more than 8100 acres of riparian, marsh, and backwater habitat along the LCRB from Lake Mead (below Hoover Dam) to the international border with Mexico (Stromberg et al., 2007; USBR, 2011). Active revegetation areas have been identified along tributaries (Virgin River, Muddy River, and Las Vegas Wash) of Lake Mead and shoreline areas along Lake Mohave.

Revegetation activities in LCRB are primarily being conducted for habitat restoration purposes by utilizing native riparian species and replacing invasive or exotic species, and as a compulsory mitigation requirement to fulfill state and federal regulations for discharging wastewater. The revegetation activities in LCR often utilize cottonwood and willow species. Other potential native riparian species for revegetation purposes includes brittlebush, creosote bush, salt heliotrope, desert marigold, globe mallow, alkali sacaton, wolfberry, iodine bush, mesquites, quail bush, and salt grass (Eckberg and Shanahan, 2009). Among these riparian species, cottonwood and willow species have been well studied for their role on soil reinforcement and improving slope stability (Simon and Collison, 2002; Pollen and Simon, 2005; Pollen-Bankhead et al., 2009; Comino and Marengo, 2010; Hubble et al., 2010). However, the mechanical functions of native riparian shrub species have been less studied in the LCRB riparian areas.Along with its broader ecological benefits, vegetation has long been recognized by river managers as increasing streambank stability (Thorne, 1990; Simon and Darby, 1999). There is a growing recognition of the important influence exerted by stream flows and channel processes on riparian vegetation structure and composition (Pollen-Bankhead et al., 2009). Yet, there is very little information on root characteristics of xeoriparian species for bank stabilization (Simon and Collison, 2002; Simon et al., 2006; Pollen-Bankhead et al., 2009).

To understand the potential of root systems for soil stabilization, some essential parameters such as root density (RD), root length density (RLD), and root diameter must be known (Gyssels et al., 2005; De Baets et al., 2008). Roots can withstand high tension while soils, on the other hand, are strong in compression and weak in tension (Simon and Collison, 2002; De Baets et al., 2008). Root permeated soil results in a reinforced soil structure with increased soil shear strength (Greenway, 1987; Thorne, 1990; Simon and Collison, 2002). Roots generally extend perpendicular to the soil surface and reinforce the soil by increasing the shear strength of the soil mass on the sheared surface; this is also known as root tensile strength (Wu et al., 1979; Reubens et al., 2007). The magnitude of root cohesion is highly dependent on root morphology since a large number of fine roots are known for higher tensile strength and contribute more to soil reinforcement in comparison to coarse roots (Operstein and Frydman, 2000; De Baets et al., 2008). Previous research has revealed a non linear inverse relationship between root diameter and root tensile strength, with smaller roots having more strength per unit root area. This also suggest that a large number of strong roots in grasses and shrubs will lead to maximum root cohesion and contribute significantly to slope stabilization (Abernethy and Rutherfurd, 2001; Simon and Collison, 2002; Pollen and Simon, 2005; De Baets et al., 2008). Many research studies have estimated root cohesion for a variety of riparian tree and herbaceous species (Abernethy and Rutherfurd, 2001; Simon and Collison, 2002; Pollen and Simon, 2005; Simon et al., 2006; Pollen-Bankhead et al., 2009), but substantially less work has been carried out to understand the root cohesion values of riparian shrub species for streambank stabilization.

The present study aims at quantifying the root cohesion of four native shrub species *A. lentiformis* (Torr.) S. Watson (Quail bush), *L. andersonii* A. Gray (Wolfberry), *L. tridentata* (DC.) Coville (Creosote bush), and *A. occidentalis* (S. Watson) Kuntze (Iodine bush) from a desert ecosystem to understand their suitability in revegetation for streambank stabilization. Field experiments were carried out in the Virgin River corridor, an analog site for LCRB. The root cohesion values generated from the experimental study can be applied for revegetation purposes in similar riparian areas at LCRB channel and its tributaries.

4.2 Materials and methods

4.2.1 Study Area

The root systems of four desert riparian shrub species were studied at the floodplain zone of the Virgin River (114^{0}16'07" W, 36^{0}41'12" N), located about 130 km northeast of the city of Las Vegas, Nevada. The climate is semi-arid with average annual precipitation of 178mm and the potential evapotranspiration rate is nearly 2400 mm (Shevenell, 1996). The soil texture in the sampling site is dominated by fine sand texture.

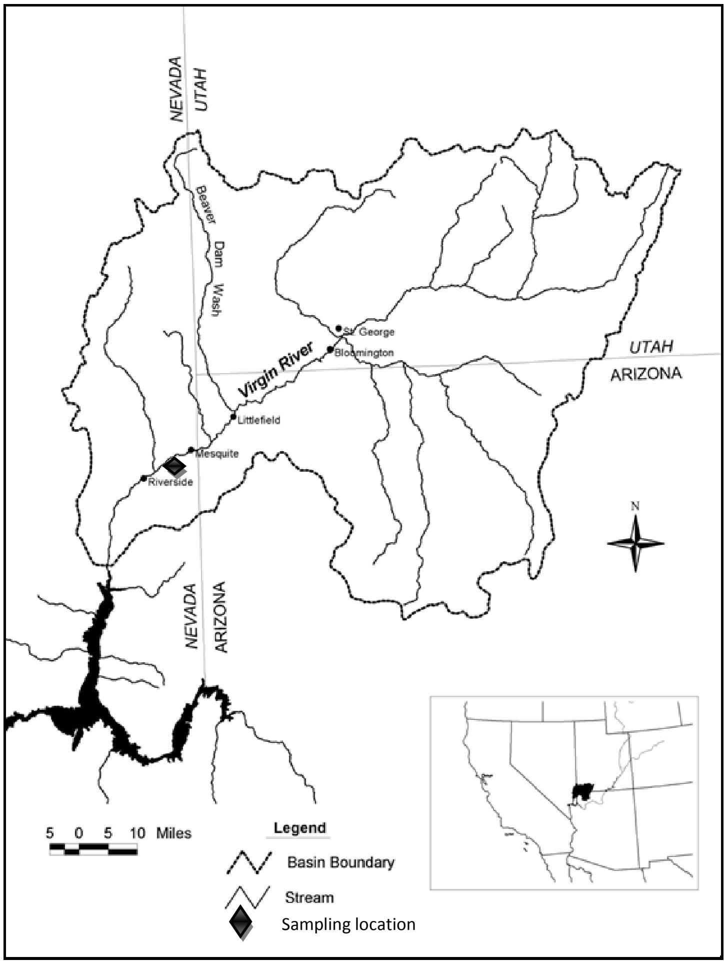

Figure 4.1 Map showing the plant sampling locations at the Lower Virgin River Watershed (Pahl, 2001)

The dynamic hydrologic system of the Virgin River supports a unique ecosystem including native flora and fauna. Three of the selected four species including *A. lentiformis, A. occidentalis*, and *L. andersonii* were generally found near the river channel but *L. tridentata* was only found at the upper reach of the floodplain area. All four species were sampled near riverside areas of the Lower Virgin River (Figure 4.1) based on the abundance of the species throughout the floodplain area. Plant species were sampled in spring 2011 (February, March, April).

4.2.2 Sampling procedure

The distribution of RD and RLD was estimated from sampling species following the dry excavation method described by Bohm (1979). Relatively young plants were selected for the present study and the dry excavation depth ranged from 0.5 to 0.8 m. The dry excavation area was defined by a contour, which was delineated at a distance from the plant stem equaling the orthogonal projected radius of the above ground biomass (De Baets et al., 2007). A soil column was then dug following 0.10 m depth intervals around the orthogonal projection passing through the root system as deep as possible. The roots from each depth interval were divided into five diameter classes representing very fine roots (<0-0.5 mm), fine roots (0.5-2 mm), and coarse roots (2-4 mm, 4-6 mm, and 6-8 mm). Very fine root samples (0-0.5 mm) were collected by using representative core sampling (approximately 240 cm^3) from different soil depths where roots were available (Bohm, 1979). Fine roots (0.5-2 mm) as well as coarse roots (> 2 mm) passing laterally through the exposed soil columns were labeled in each soil depth. The labeled roots from each soil depth were cut and directly measured for diameter and length, and then oven-dried at 70 °C for 48 hours and weighed. The above-ground parts of sampling trees were oven-dried at 70 °C for 48 hours and weighed. The very fine roots (< 0.5 mm) were extracted by sieving the soil samples with 0.5 mm sieve and preserved with 30% ethanol in the laboratory. Root length and diameter were directly measured for RD and RLD analysis. The average RD (kg m^{-3}) for each soil depth and individual species was estimated by dividing the mean dry mass of the roots by the volume of the soil cylinder for each depth class (De Baets et al., 2007).

$$RD = \frac{M_D}{V} \qquad (1)$$

$$RLD = \frac{L_R}{V} \qquad (2)$$

Where M_D (kg) is the mean dry root mass and V (m^3) is the volume of the corresponding soil cylinder. The soil volume was calculated referring the mean diameter of the orthogonal projection of the above ground biomass and the maximum root depth considered for each species. The calculation of RD and RLD for each depth class of 0.10 m, soil volume was divided by the number of depth classes. Average RLD (km m^{-3}) for each depth class and individual species was estimated by dividing the mean root length (L_R, km) by the volume (V, m^3) of the root permeated soil sample (Smit et al., 2000; De Baets et al., 2007). A root diameter range of 0-8 mm was considered in this study since previous work has shown that larger roots do not contribute significantly to increase soil strength (Burroughs and Thomas, 1977; Ziemer, 1981). Larger roots have an important function as individual anchors rather than a component of soil strength (Coppin and Richards, 1990).

4.2.3 Root area ratio

Root area ratio is a fraction of soil cross-sectional area occupied by roots per unit area and is used to estimate the root contribution to soil strength (Gray and Leiser, 1982; De Baets et al., 2008). The RARs were estimated using the root diameter (> 0.5 mm) and RLD information from each soil depth among all the studied species. For each species, RAR was estimated at different depths by measuring the total length of roots having similar diameters and measuring their mean cross-sectional area. The total length per diameter class was divided by 0.10 m to obtain the number of 10 cm long root segments for all roots and measuring their mean cross-sectional area as described in De Baets et al. (2008).

$$RAR = \frac{\sum_i^n n_i a_i}{A} \qquad (3)$$

Where, n_i is the number of 10 cm long roots in each root diameter class and a_i is mean root cross sectional area of a root diameter class (m^2) and A is the horizontal cross-sectional area or reference area (m^2) determined by the vertical orthogonal projection of the above-ground biomass of the plant. In Eq. 3, the total number of roots will be overestimated, but their mean cross-sectional area at a certain depth will be underestimated, because when roots cross under a certain angle their cross-sectional area will be larger. Therefore, if the assumption is made that the roots are growing vertically, one can argue that overestimation of the number of roots will be somewhat compensated by the underestimation of root cross-sectional areas (De Baets et al., 2008).

The RAR values for the very fine roots (< 0.5 mm) were analyzed through representative soil core sampling in each soil depth. The RAR was calculated from scanned images of very fine roots (<0.5 mm) by using ImageJ software (Abràmoff et al., 2004).

4.2.4 Root tensile strength

Root tensile strength was measured in situ by pulling on roots that were exposed on a trench wall. The trench was excavated outside the wall of radial orthogonal projection of the above-ground biomass. The excavation inside the projected wall was made using hand tools, and several roots sizes (0.5-8 mm diameter) from different depth classes were exposed for testing. An in situ root tensile testing devise (jig) was fabricated for the root tensile strength measurements as described in Abernethy and Rutherfurd (2001). The root pulling devise consisted of a bearing plate with the center removed for access to the roots and with four legs that extended back to a hand operated boat winch (max load 900 kg). The pulling devise was positioned against the trench wall to conduct the tensile strength tests. A load cell was attached to the boat winch at one end and another end was attached to a clamping jaw. Different sizes of jaws were applied to clamp the roots of various diameters. An average length of 15 cm from the trench wall was maintained for root pulling to avoid slippage during shearing process. The load cell was connected to a data logger (Campbell scientific CR1000) that recorded the applied tensile force every second. The maximum load was applied until the root failure and the root diameter at the point of rupture was measured. Root pulling was carried out for all root depths and available root diameter classes for each species. For more details on the root pulling devise, consult Abernethy and Rutherfurd (2001) and Tosi (2007).

The root pulling test for the very fine root diameter class (0-0.5 mm) was performed using a manual dynamometer (maximum 9 kg force) and small size clamping jaw. One end of the dynamometer was attached with a clamping jaw and the other side was used for manual pull. The root failure was noted from the dynamometer reading for each root specimen. Both load cell and dynamometer were calibrated by applying a known force in the laboratory and were also verified with trial root pulling tests for consistent results. The relationship between tensile strength and root diameter can be explained by using a power law equation ($T_r = aD^{-b}$) for the species tested in this study, where a and b are empirical constants depending on species.

4.2.5 Root cohesion estimation

The root reinforcement model of Wu et al. (1979) was used to estimate the increase in soil shear strength due to the presence of roots. It is a modified version of the simple perpendicular root model developed by Waldron and Dakessian (1981). The model assumes that all roots grow vertically and act as loaded piles such that tension is transferred to them as the soil is sheared (De Baets et al., 2008). The assumption that fibers are oriented perpendicular to the shear plane is useful because it yields an average estimate of all possible orientations (Gray and Sotir, 1996). The plant roots tend to bind the soil together in a monolithic mass and contribute to the strength by providing an apparent additional cohesion (Abernethy and Rutherford, 2001). The increased soil shear strength by root can be expressed as an additional cohesion

$$S_r = S + C_r \qquad (4)$$

Where S is soil shear strength (kPa), S_r (kPa) is the shear strength of the soil reinforced by roots and C_r (kPa) is the increase in shear strength due to the presence of roots. The shear force is responsible for fiber deformation and fiber stretch; it provided sufficient interface friction, confining stress and anchorage length to lock the fiber in place and to prevent slippage or pullout (De Baets et al., 2008). The tension developed in the roots as the soil sheared is estimated with a tangential component resisting shear and a normal component increasing the confining pressure on the shear plane. The major critical assumption of this model is that all roots attain ultimate tensile strength simultaneously during soil shearing (Pollen et al., 2005). The increase in shear from a full mobilization of root tensile is represented by:

$$C_r = t_R(sin\theta + cos\theta\ tan\text{Ø}) \qquad (5)$$

Where Θ is the angle of shear distortion in the shear zone, Ø is the soil frictional angle and t_R is the total mobilized tensile stress of root fibers per unit area of soil (Waldron, 1977; Wu et al., 1979). Equation 5 further requires the tensile strength of roots and RAR. The angle of the internal friction of the soil is found to be affected by the presence of roots (Gray, 1974). The sensitivity analyses of the Wu et al. (1979) showed that the value of the bracketed term in equation (5) is fairly insensitive to normal variations in Ø and Θ (40-90^0 and 25-40^0, respectively) with values ranging from 1.0 to 1.3. The average value of 1.2 was selected by Wu et al. (1979) to simplify equation (5). Considering the fine sand texture of the study area, a friction angle of 27^0 was selected (Thomas and Pollen-Bankhead, 2010). The angle of shear distortion is assumed to be 45^0. Using these values, the bracketed term of Eq. (5) equals 1.06. Thus, the equation used in this study to calculate root cohesion in sandy soil becomes:

$$C_r = 1.06 \frac{\sum T_i n_i \alpha_i}{A} \qquad (6)$$

Where T_i is root tensile strength (MPa), n_i is the number of roots in a diameter class, i is root diameter class, α_i is the root cross-sectional area (m^2) and A is the reference area of soil occupied by roots (m^2). The Wu's perpendicular model assumes that all roots crossing the shear plane break at the same time. However the driving forces exerted on the soil surface are not sufficient to break all roots, which results in an overestimation of root strength (Pollen and Simon, 2005). Therefore, the estimated root cohesion values based on the Wu's model are maximum values, which can be only useful to rank species according to their soil reinforcement potential.

Pollen and Simon (2005) estimated root cohesion using the fiber bundle model (FBM), which was developed to correct for overestimation made by applying Wu et al. (1979)'s model. The FBM takes into account the fact that roots within the soil matrix have different maximum strengths, and therefore break at different points as a load is applied to the soil. Also this model redistributes the load from the broken roots to the remaining intact roots crossing the shear surface (Pollen and Simon, 2005). Root reinforcement model (Rip Root) was applied in the present work, which uses the fiber bundle theory to improve the estimates of root reinforcement (Simon et al., 2010) and compared to the perpendicular root model of Wu et al. (1979). The root size >0.5 mm was considered for the estimation of root cohesion using FBM, since number and size of the exposed roots were measured at each depth profile. In some studies the role of very fine roots (0-0.5 mm) has been questioned due to their rapid turnover. Additionally, their length could be less reliable to avoid slippage during shearing process (Bischetti et al., 2009). However, the cohesion values obtained from the soil depths > 0.50 m are not considered due to fewer roots available for FBM run.

4.2.6 Data analysis

Data analyses were carried out using JMP software (SAS Institute, Cary, North Carolina). A Kolmogorov-Smirnov test was used to test the normality of the data before proceeding with analyses of variance. One-way analysis of variance (ANOVA) was used to study the observed variability in root distribution and root tensile strength among the four shrub species. Differences detected by ANOVA for RD, RLD, RAR, and root cohesion at two different soil depths (0-0.3 m and >0.3 m) were compared among each species using *p* values from the Kruskal-Wallis test, which was used since the dependent variables were not normally distributed. This comparison between < 0.3 m and > 0.3 m is relevant since the root distribution in the top 0.30 m of the soil is known to be important for the soil's resistance against concentrated flow erosion (Gyssels et al., 2005; De Baets et al., 2007).The power law equations were fitted to explain tensile strength and root diameter relationships and evaluated based on adjusted R^2 values and significance of the parameters *a* and *b* from the power law equation. Since the root diameter sizes varied in the present study, ANCOVA was applied to detect the differences in root tensile strength among the species.

4.3 Results

General morphological characteristics of the plant species used in the study are summarized in Table 4.1. The tree ring observations showed that the ages of the four shrub samples ranged between 2-4 years. The above-ground dry biomass (gram/shrub species) measured for *A. lentiformis* (460 ± 135), *L. andersonii* (206 ± 29), *L. tridentata* (288.6 ± 92), and *A. occidentalis* (203 ± 16).

Table 4.1 Morphological characteristic of sampled shrub species in the Lower Virgin River

Name of Species	Common Name	(N)	H (m)	D_{sv} (m)	d_{max} (m)	DB_{ag} (g)
A. lentiformis	Quail bush	3	0.65±0.15	0.93±0.08	0.8	460±135
L. andersonii	Wolfberry	2	0.82±0.18	0.80±0.01	0.5	206± 29
L. tridentata	Creosote bush	2	1.1±0.21	0.9±0.06	0.8	288±92
A. occidentalis	Iodine bush	2	0.41±0.12	0.80±0.02	0.7	203±16

N = Number of samples, H = Average plant height (meter), D_{sv}= Average diameter of rooted soil volume (meter), d_{max}= Maximum depth for the plant root sampling (meter), DB_{ag} = Average dry above-ground biomass (gram).

The variation among above-ground biomass for the four species might be the result of phenological variation, since the field study was carried out during early spring (February-March 2011). All studied plant species have tap root systems with branches extending to different maximum soil depths (0.5-0.8 m).

4.3.1 Root density and root length density

The average RD and RLD among the four species were analyzed at different soil depths (Figure 4.2a and 4.2b). Root distributions showed a general trend of increased RD in the first 0-0.5 m of soil and then decreased with increasing soil depth. Since RD is the result of total root dry biomass per unit volume of soil, coarse root (>2mm diameter) biomass largely determines the RD at various soil depths for the studied species. Among the four shrub species *A. lentiformis* has the maximum RD value at 0.40 m (0.52 kg m^{-3}) followed by *L. andersonii* at 0.30 m (0.42 kg m^{-3}), *A. occidentalis* at 0.20 m (0.40 kg m^{-3}), and *L. tridentata* at 0.50 m (0.39 kg m^{-3}) (Figure 4.2a).

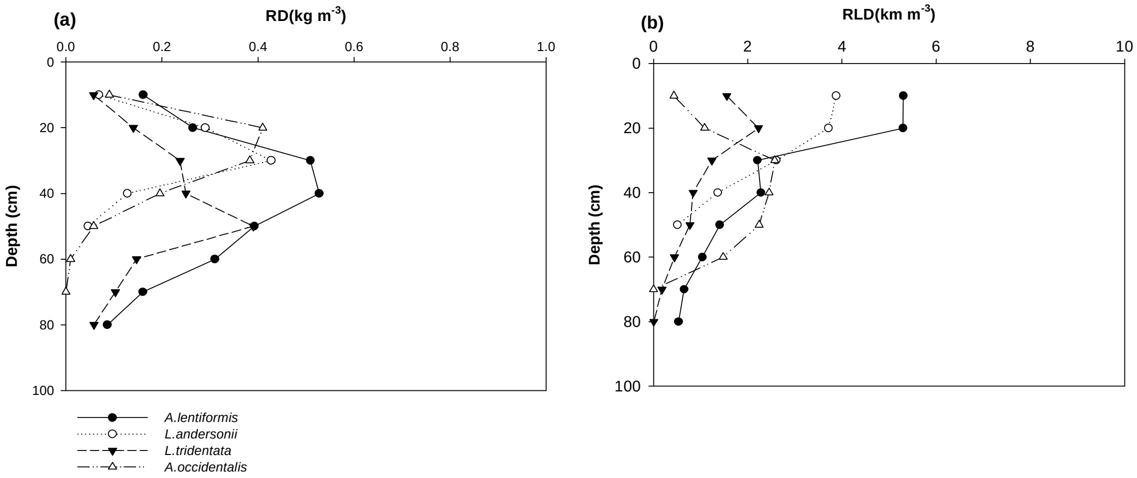

Figure 4.2 The distribution of a) root density (kg m^{-3}) and b) root length density (km m^{-3}) at different soil depths for four shrub species. (x- axis values are different for root density and root length density)

The RLD values among the four shrub species peaked in the first 0-0.3 m soil and decreased with increasing soil depth (Figure 4.2b). The maximum RLD estimates, in decreasing order, equaled 5.3 km m^{-3} for *A. lentiformis,* 3.87 km m^{-3} for *L. andersonii*, 2.57 km m^{-3} for *A. occidentalis*, and 2.23 km m^{-3} for *L. tridentata* (Figure 4.2b). The RLD for topsoil (0-0.3 m) was found to be significantly higher ($p < 0.01$) in comparison to the subsoil (>0.3 m).

4.3.2 Root area ratio

Root area ratio was calculated for each 0.10 m soil layer and compared among the plant species using the mean values over all measured depths. The mean RAR for *A. lentiformis* (0.0077) tended to be higher, followed by *L. andersonii* (0.0038), *A. occidentalis* (0.0036) and *L. tridentata* (0.0033) (Figure 4.3). The change in root distribution with depth for the different root diameter classes varied among the four species (Figure 4.3). The maximum RAR was detected at 0.4 m soil depth among the studied species except for *L. tridentata*, for which the maximum RAR was estimated at 0.5 m soil depth. In general, a trend of increasing RAR with depth was noted among all species, similar to the trend described for RD (Figure 4.3).

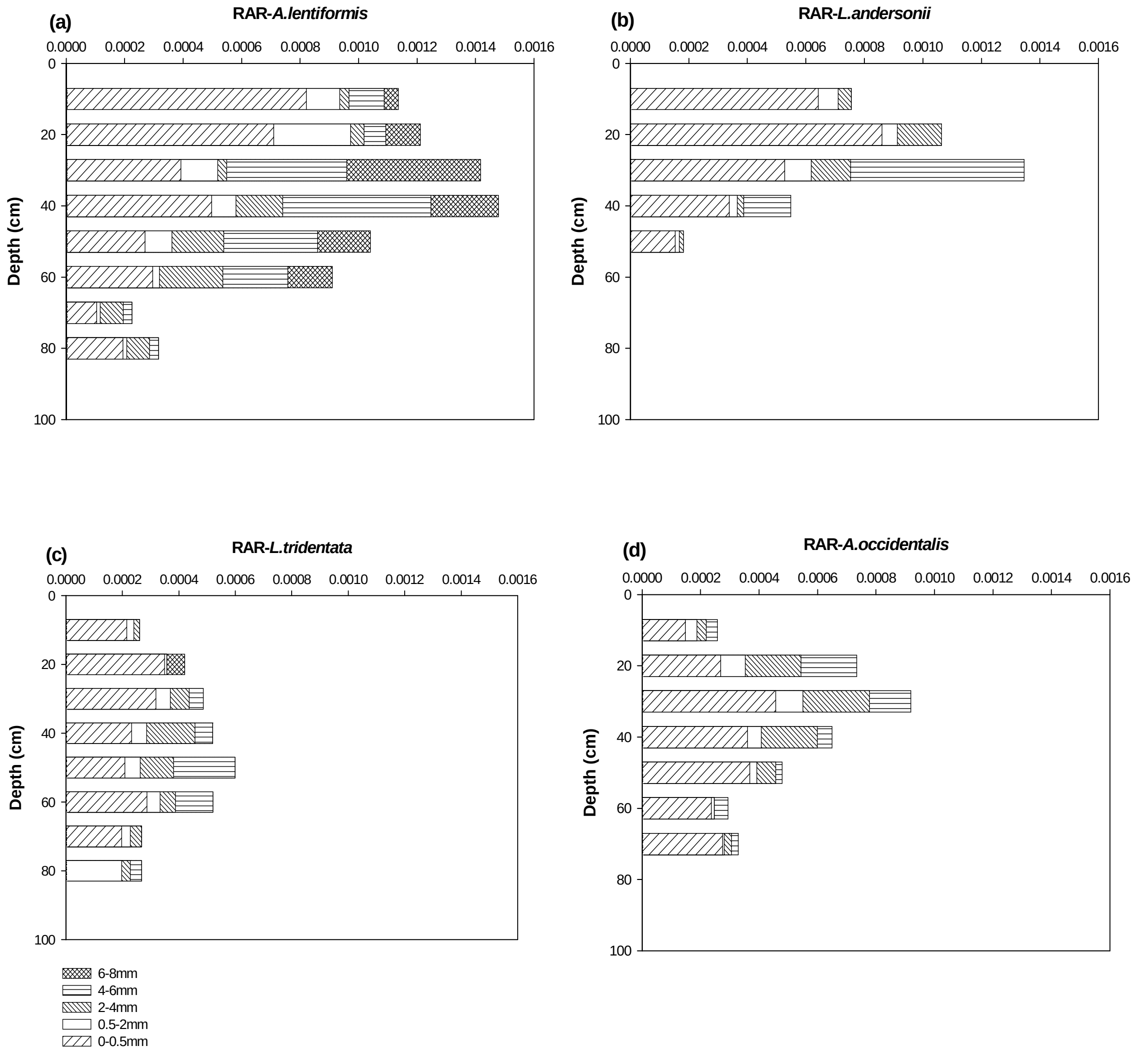

Figure 4.3 Distribution of root area ratio (RAR) with depth, subdivided by root diameter for a) *A. lentiformis*, b) *L. andersonii*, c) *L. tridentata*, and d) *A. occidentalis*

The contribution of the very fine root diameter class, 0-0.5 mm, to RAR was greatest among all root diameter classes for all species. No consistent trends were observed on the RAR contribution by coarse root diameters (>2-8 mm) among the four species. The fine root RAR distribution was significantly higher ($p < 0.01$) for soil depth 0-0.3 m in comparison to the >0.3 m (Figure 4.4a). There was no significant relationship observed from the RAR distribution of coarse roots among different species and soil depths (Figure 4.4b).

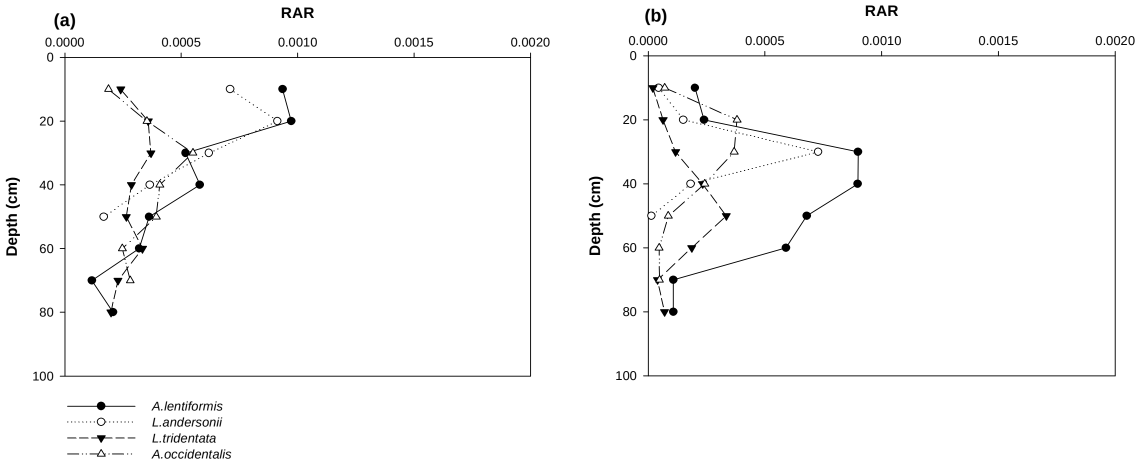

Figure 4.4 Root area ratio (RAR) distribution for a) fine roots (D = 0-2mm) and b) coarse roots (D = >2mm) at various soil depths (0-0.8 m) for four shrub species

4.3.3 Tensile strength

In situ root tensile strength tests were performed on the four native shrub species. Sample number, root diameter, and parameter values for the established power law relationships are listed in Table 4.2. The root tensile strength was calculated for every 0.10 m soil depth and different root diameter classes, also the average tensile strength value for individual species was calculated and compared among the four plant species. The mean root tensile strength (average values at all soil depths) varied among four species. Among the four shrub species, *L. tridentata* (62.23 MPa) showed the highest value of root tensile strength followed by *L. andersonii* (53.53 MPa), *A. lentiformis* (49.17 MPa), and *A. occidentalis* (35.03 MPa) (Table 4.2). The comparison of root tensile strength and root diameter classes showed a non linear and inverse relationship similar as reported by previous studies (Simon and Collison, 2002; Bischetti et al., 2007; Tosi, 2007; De Baets et al., 2008; Comino and Marengo, 2010). Tensile strength values for the different diameter classes were fitted into power law equations to characterize the relative strength of the root system for the four species (Figure 4.5).

Table 4.2 Diameter range for four shrub species and parameter values obtained from the power law equations showing the inverse relationship between root diameter and root tensile strength.

Plant Species	Diameter range (mm)	Mean T_r (MPa)	*a*	*b*	*n*	Adjusted R^2
A. lentiformis	0.20-6.5	49.17± 7.01	31.38	-0.89	46	0.82
L. andersonii	0.20-5.5	53.53±6.78	30.54	-0.82	52	0.7
L. tridentata	0.20-2.30	62.23± 4.76	42.34	-0.68	44	0.59
A. occidentalis	0.20-3.8	35.03±2.47	29.17	-0.62	54	0.81

Mean values, ± standard error, *a* and *b* = parameter values for power law equation, *n*= number of roots tested per species.

The mean root tensile strength for *L. tridentata* species was significantly different from A. *lentiformis* ($p < 0.02$), *L. andersonii* ($p < 0.01$), and *A. occidentalis* ($p < 0.01$) (Figure 4.5). The mean tensile strength for the root diameter class (0-0.5 mm) was also significantly different ($p < 0.01$) among four species. The maximum root tensile strength for the 0-5 mm root diameter class was 201.6 MPa for *L. andersonii*, 159.1 MPa for *A. lentiformis*, 119.3 MPa for *L. tridentata*, and 95.4 MPa for *A. occidentalis* species. This significant difference could be the result of proportionally a greater cohesive strength provided by smaller diameter roots than larger as explained in power law. However, the mean root tensile strengths for the root diameter >0.5 mm were not significantly different among four species. The discrepancy in root tensile strengths among species between small (<0.5 mm) and large root diameter (>0.5 mm) classes could be ascribed to uncertainty in diameter measurement and in autocorrelation between diameter and root strength (Hales et al., 2009). The power regression between *T*r and root diameter (Table 4.2) were fitted for ANCOVA application (McDonald, 2009) and checked the parallelism between the fitted lines through exponential parameter '*b*'. The null hypothesis of the same slope was rejected (F = 5.6, p = 0.001), and ANCOVA was not persuaded for further analysis.

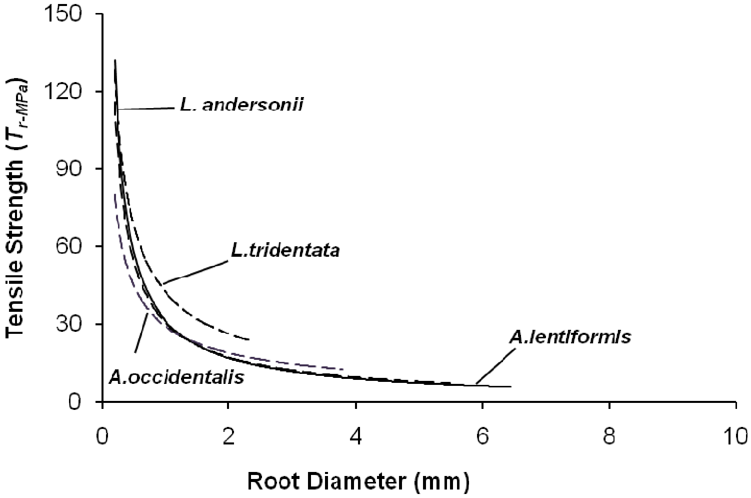

Figure 4.5 Power law relationship between root diameter (mm) and tensile strength (MPa) for *A. lentiformis*, *L. andersonii*, *L. tridentata*, and *A. occidentalis* from in situ tensile strength test

4.3.4 Root cohesion

Root cohesion in the present study was calculated following Wu's model, which was broadly applied in previous studies (Simon and Collison, 2002; Pollen and Simon, 2005; Tosi et al., 2007; De Baets et al., 2008; Bischetti et al., 2009; Preti and Giadrossich, 2009; Hubble et al., 2010; Comino and Druetta, 2010; Comino and Marengo, 2010). Root cohesion is the product of mean root tensile strength and the RAR distribution (Eq. 5). The average root cohesion values for the entire soil profile were 56.3, 54.7, 24.5, and 21.4 kPa for *A. lentiformis*, *L. andersonii*, *L. tridentata*, and *A. occidentalis*, respectively. The additional cohesion provided by roots of each species was also estimated for each soil depth. The maximum root cohesion was observed in the topsoil (0-0.3 m) for all the plant species. The maximum additional cohesion was found at the 0.1 m soil depth for *A. lentiformis* (97.68 kPa), at the 0.2 m depth class for *L. andersonii* (89.3 kPa) and *L. tridentata* (35.6 kPa), and at the 0.3 m depth class for *A. occidentalis* (34.8 kPa) (Figure 4.6).

Despite the higher mean root tensile strength of *L. tridentata*, its root cohesion was less than that of A. *lentiformis* and *L. andersonii*. This observation suggests that the root cohesion value was consistent with the pattern observed in RAR distribution among all studied species. It was also evident that the fine root class was the dominant source of root cohesion (Figure 4.6). The maximum root cohesion (kPa) was found in the 0-0.3 m topsoil, where the dense fine root distribution resulted in the high RAR values. The root cohesion contributed by fine roots was significantly higher (p <0.01) in both soil depths (0-0.3 m and >0.3 m) than the root cohesion by coarse roots among all species. Also, the fine root contribution to soil cohesion was significantly higher (p < 0.03) in the 0-0.3 m topsoil compared to the deeper soil among the four shrub species (Figure 4.7a). However the root cohesion provided by the coarse roots (>2-8mm) did not significantly differ between topsoil (0-0.3 m) and subsoil (>0.3 m).

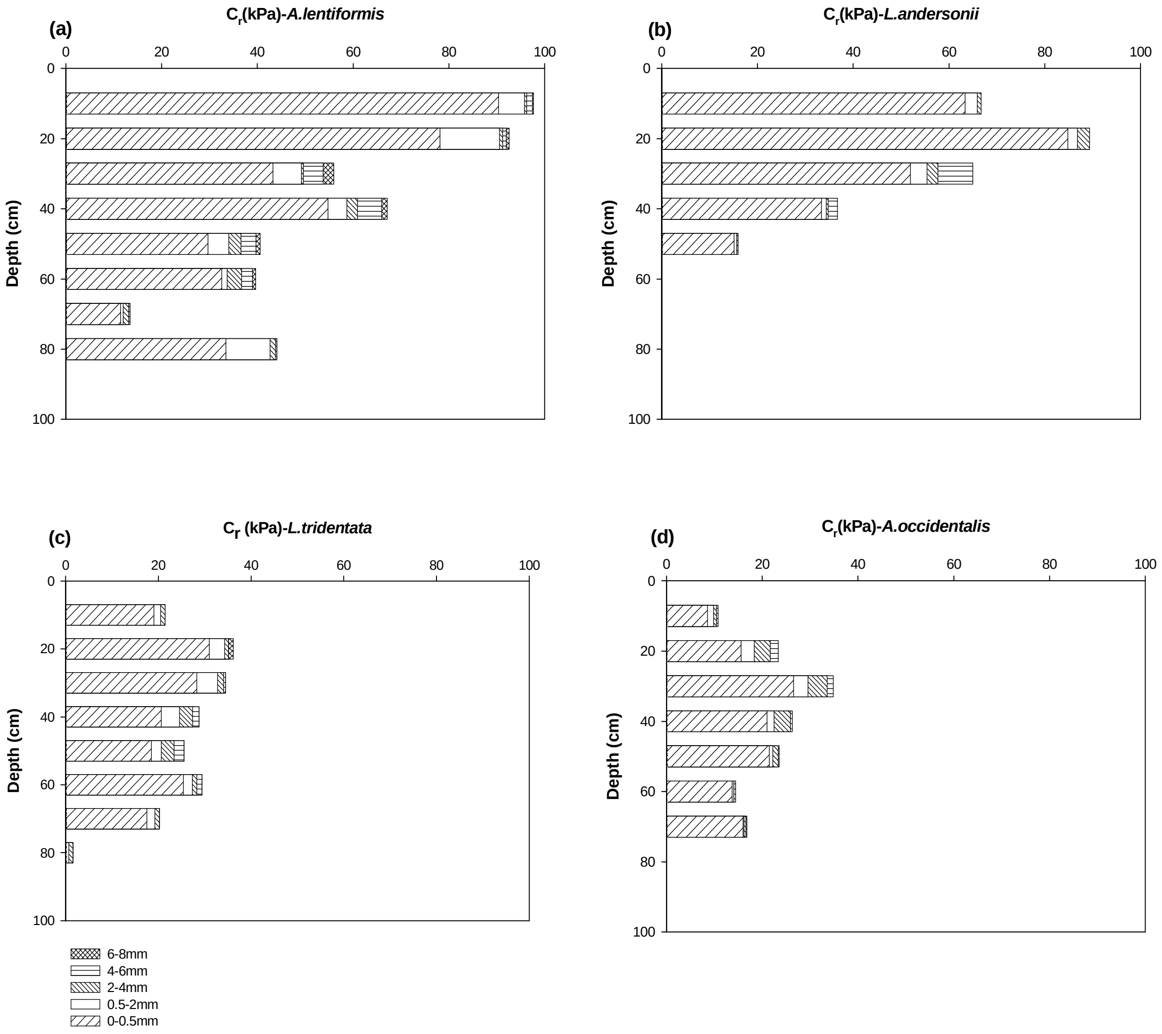

Figure 4.6 Distribution of root cohesion (C_r kPa) due to roots with depth, subdivided by root diameter for a) *A. lentiformis*, b) *L. andersonii*, c) *L. tridentata*, and d) *A. occidentalis*

The root cohesion values estimated by Wu's model (C_r) and Fiber bundle model (C_{FBM}) are presented in Table 4.3. The reinforcement effect among different species varies for different soil depths; however, both models' outcomes followed the same trend of higher values within 0.2-0.5 m and lower with increasing depth. The highest C_r value was estimated for *A. lentiformis* (5.06 kPa at 0.20 m), followed by *L. andersonii* (2.83 kPa at 0.30 m), *L. tridentata* (2.69 kPa at 0.50 m), and *A. occidentalis* (2.56 kPa at 0.20 m) respectively. Reduction factor was estimated by taking the ratio between the C_{FBM} and the C_r estimates as defined in Bischetti et al. (2009), and presented in Table 4.3.

Table 4.3 Root cohesion values for four desert shrub species and ratio between Wu's model and Fiber Bundle Model for root diameter > 0.5 mm

Depth (m)	***A. lentiformis***			***L. andersonii***			***L. tridentata***			***A. occidentalis***		
	C_r kPa	C_{FBM} kPa	C_{FBM}/C_r	C_r kPa	C_{FBM} kPa	C_{FBM}/C_r	C_r kPa	C_{FBM} kPa	C_{FBM}/C_r	C_r kPa	C_{FBM} kPa	C_{FBM}/C_r
0.1	4.9	2.6	0.5	1.6	0.8	0.5	1.1	0.5	0.4	1.3	0.5	0.3
0.2	9.0	5.0	0.5	4.4	1.8	0.4	2.8	1.0	0.3	6.7	2.5	0.3
0.3	8.7	3.5	0.4	7.0	2.8	0.4	4.0	1.6	0.4	5.6	2.2	0.4
0.4	9.2	3.5	0.3	1.7	0.8	0.4	6.0	2.3	0.3	3.7	1.9	0.5
0.5	7.9	2.7	0.3	0.2	0.1	0.5	6.6	2.6	0.4	1.6	0.7	0.4

C_r= root cohesion values obtained using Wu's model; C_{FBM} = root cohesion values obtained using FBM model; C_{FBM}/C_r = ratio between C_{FBM} and C_r.

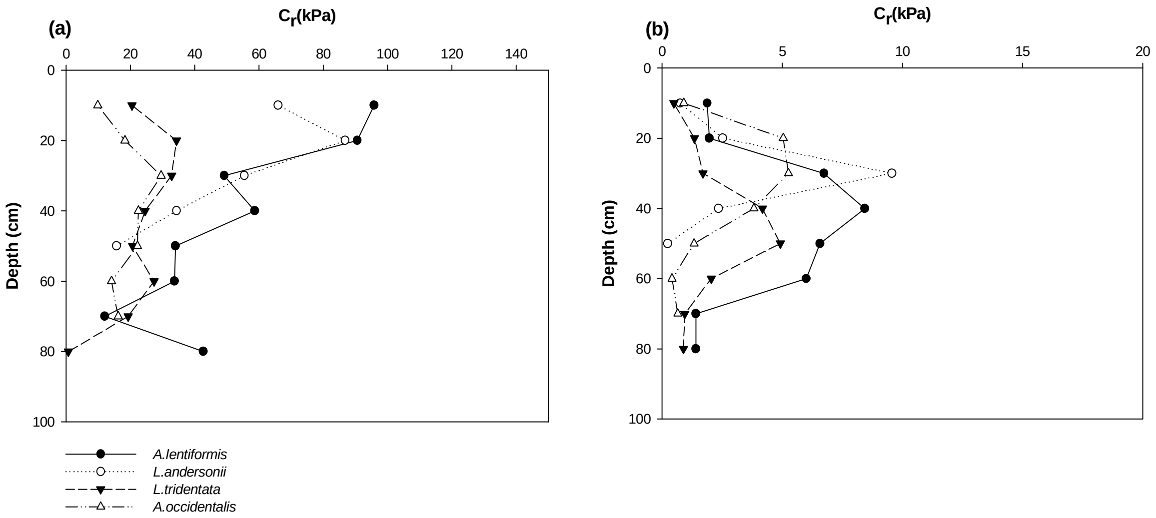

Figure 4.7 Root cohesion (C_r kPa) distribution for a) fine roots (D = 0-2mm) and b) coarse roots (D >2mm) at various soil depth (0-0.9 m) for four shrub species (x-axis values are different for fine roots and coarse roots)

4.4 Discussion

4.4.1 Root length and root length density

The RD range for different soil depth classes in the present study (0.04-0.52 kg m^{-3}) stands at the lower end of the RD values measured for the native Mediterranean plant species grown in ephemeral channels of loamy deposits (De Baets et al., 2007). De Baets et al. (2007) measured the RD (0.1-7 kg m^{-3}) for native shrubs *Dittrichia viscosa*, *Artiplex halimus*, *Retama sphaerocarpa*, and *Nerium oleander* in the Cárcavo catchment in Southeast Spain, and observed higher RD at the topsoil with a gradual decrease of RD with increasing soil depth. The different trends of RD distribution observed for topsoil vs. subsoil (0-0.3 m and >0.3 m) in this study are consistent with the observation made by Schenk and Jackson (2000*b*). Earlier, Wilcox et al. (2004) studied four shrub species from the Mojave Desert, and found different root foraging strategies and niches within the soil profile; but similar to this study, they observed highest number of roots between 0.2 and 0.4 m.

The high density of very fine roots (0-0.5 mm) and fine roots (0.5-2 mm) in the topsoil (0-0.3 m) resulted in high RLD estimate in our study. It is probably linked to the higher availability of soil water in the topsoil layers in comparison to the subsoil layers. In a study by De Baets et al. (2007), a high density of fine roots in the topsoil was observed among plant species growing in ephemeral channels, where soil water availability was higher as compared to abandoned croplands or badland slopes. Based on the root distribution results, *A. lentiformis* and *L. andersonii* could have a relatively higher erosion reducing potential due to the high density of fine roots near the soil surface (0.3 m) compared to *A. occidentalis* and *L. tridentata*. In a comparative study of effectiveness of shoot against roots for *Rosmarinus officinalis*, *Stripa tenacissima*, and *Anthyllis* species in the Mediterranean region, Bochet et al. (2006) found a greater role of shoot for preventing splash and interrill erosion. Based on that, *A. lentiformis* could be the most effective in reducing interrill soil loss, since it has the largest crown cover among four species (Table 4.1). Moreover, since *A. lentiformis* has a high RD, it could be effective for reducing concentrated flow erosion rates as well (De Baets et al., 2007).

4.4.2 Root area ratio

Generally the root distribution in temperate climates had a decreasing trend of RAR with increasing soil depth (De Baets et al., 2008), whereas this study shows first an increase of RAR which is then followed by a decrease. In a study of the global biogeography of roots, Schenk and Jackson (2002*b*) reported lower root densities in the upper 20 cm of the soil profile than in the interval from 20 cm to 40 cm. More than four-fifth of these results were from the arid ecosystem where the upper soil horizons are likely to be too dry for resource uptake during some parts of the growing season. From another root system study in water-limited ecosystems, Schenk and Jackson (2002*a*) reported high root densities at a greater depth. It is due to plants' tendency in water-limited ecosystems to access water from deeper soil layers, in which water is stored from occasional and seasonal wet periods (Schenk and Jackson, 2002*a*). It is to be noted that large diameter roots (>8 mm) were not considered in this study, which might contribute differently to root distribution (RD and RAR).

From the RAR analysis, it is known that native shrub species' roots occupy approximately 0.46 % of the area under the crown of the plants. Mattia et al. (2005) and De Baets et al. (2008) obtained RAR values equaling less than 1% of the area under the crown for Mediterranean plants. Similar RAR values were obtained by Abernethy and Rutherfurd (2001) and Simon and Collison (2002) for riparian tree species including river birch, black willow, sweetgum, sycamore, swamp paperbark, and river red gum for the top 1 m of soil. The RAR values of riparian trees varied between 0.01% and 0.75% with individual root diameters varying between 0.5 mm and 20 mm (Hubble et al., 2010). The present study found that *A. lentiformis* and *L. andersonii* have higher RAR values for the topsoil as compared to *A. occidentalis* and *L. tridentata*. Gibbens and Lenz (2001) studied the root systems of similar species from the Chihuahuan Desert and reported both vertical and lateral root distribution at a greater depth in contrast to the present study. This could be the result of spatial and temporal heterogeneity of soil, which might be resulted into a diverse rooting architecture (Rundel and Nobel, 1991). A number of environmental factors including soil moisture, microsite relations, nutrients, and soil temperatures could influence the root distribution to favor shallow over deep roots, suggesting that root profiles of plant communities may tend to be as shallow as possible (Pregitzer et al., 2000; Schenk and Jackson, 2002*b*; Wilcox et al., 2004; Schenk, 2008).

4.4.3 Root tensile strength

The relationships for diameter (*D*) and root tensile strength (T_r) for the four species are shown in Figure 4.5. T_r decreased with increasing *D* and followed a power law equations as reported in previous studies (Bischetti et al., 2007; Mattia et al., 2005; Tosi, 2007; De Baets et al., 2008) : $T_r = aD^{-b}$. The shape of the curves from this study emphasized the contribution of small roots, having a greater strength. The regression curves fitted in this study are comparable with other studies (Gray and Sotir, 1996; Simon and Collison, 2002; Pollen et al., 2004; Simon et al., 2006; De Baets et al., 2008). The R^2 values of the fitted power curves for *A. lentiformis,* and *A. occidentalis* were higher (above 0.8) compared to the *L. andersonii* and *L. tridentata* equaling 0.7 and 0.59, respectively ($p < 0.01$). The relationships observed between root tensile strength and root diameter for *Artiplex* species in Mattia et al.(2005) and De Baets et al. (2008) were $T_r = 72.97D^{-0.60}$ and $T_r = 45.59D^{-0.56}$ respectively. For a similar root diameter class (0-0.5 mm), *A. lentiformis* shows a different relationship than the previous studies (Table 4.2). The differences can be attributed to the different procedures for root tensile strength measurements. The observations from Mattia et al. (2005) and De Baets et al. (2008) were based on laboratory study, while in situ root tensile tests were performed in the present study.

Root tensile strength of riparian trees for 2-3 mm diameter roots estimated by Simon and Collison (2002) were lower for river birch (22 MPa), sweet gum (18 MPa), and sycamore (45 MPa) compared to the shrub species measured in this study. The tensile strength values measured by Tosi (2007) for three shrubs *Spartium junceum* (30.32 MPa), *Rosa canina* (18.91MPa), and *Inula viscose* (14.79MPa) in the Italian Apennines at similar diameter classes fit in the lower end of the present findings. For similar root diameters, tensile strengths from the present data were generally higher than the root strengths of shrub species from the Mediterranean environment and Northern Italian Apennines (Mattia et al., 2005; Tosi, 2007) . The role of very fine roots has been questioned in recent literatures due to their rapid turnover and their length that could be not enough to avoid slippage during shearing process (Stokes et al., 2009; Day et al., 2010; Schwarz et al., 2010). Also the tensile strength of the very fine roots (0-5 mm) among four species were at the high end of the values obtained for riparian trees and shrubs found in the US (Simon et al., 2006; Pollen-Bankhead et al., 2009). This difference might be the result of different methods applied, and potential error in dynamometer reading as manual dynamometer was applied in the present study for ex situ tensile strength test for very fine roots (<0.5 mm).

4.4.4 Root cohesion

The root cohesion (C_r) was estimated following the perpendicular root model developed by Wu et al. (1979). Similar to the present study observations, Abernethy and Rutherfurd (2001) and Simon and Collison (2002) reported that the RAR of vegetation is the most important factor contributing to soil shear strength. Waldron and Dakessian

(1981) and van Beek et al. (2005) further reported that fine roots will yield a larger root cohesion value than coarse roots. The force required to break a root increases linearly with increasing root diameter, but as tensile strength is calculated per unit area (Pollen and Simon, 2005), smaller roots have higher tensile strength. From the present observation, *A. lentiformis* and *L. andersonii* have exponent values of -0.89 and -0.82 compared to *L. tridentata* and *A. occidentalis* having exponent values of -0.68 and -0.62 respectively (Table 4.2). Based on the observations of Pollen and Simon (2005), the more negative the exponent of the tensile strength curve, the lower would be the overestimation for the Wu et al. equation. Root cohesion values were compared for four native shrubs from the present study with the results reported by Mattia et al. (2005) and De Baets et al. (2008) for Mediterranean shrub species including *Artiplex halimus* and *Lygeum spartum*. Root cohesion values were found to be higher than those estimated for the Mediterranean shrub species at similar soil depths (Figure 4.8). The variable plant size, methods to estimate root reinforcement, and environmental growth conditions could all explain these differences in root cohesion values (De Baets et al., 2008). The root cohesion values reported for Australian riparian species (River Red Gum, Swamp Paper bark, and Elderberry) observed by Abernethy and Rutherfurd (2001) were similar to the present observations. However the root cohesion values estimated for riparian trees by Pollen and Simon (2005) and Pollen-Bankhead et al. (2009) were lower than the ones estimated in this study.

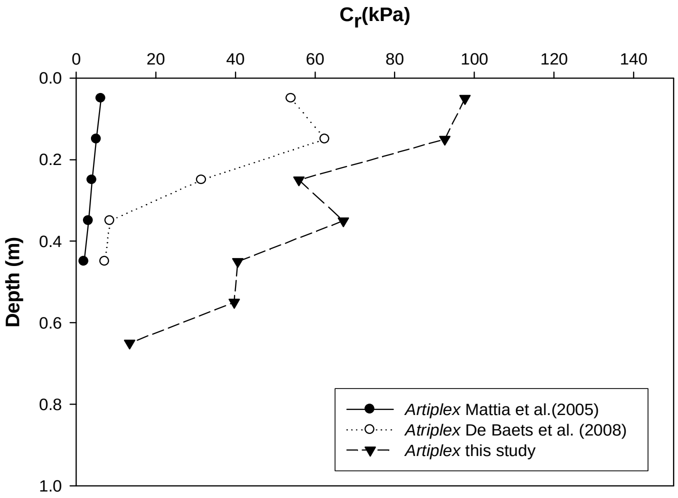

Figure 4.8 Comparison of root cohesion (C_r kPa) distribution with depth for *Artiplex* amongst different studies

Similar to the observation made by Pollen and Simon (2005), Bischetti et al. (2009), Loades et al. (2009), Comino and Marengo (2010) the results from C_{FBM} in this study also underestimates C_r results. Reduction factor for four shrub species in present study ranges between 0.35-0.56. C_{FBM} values obtained from FBM model were comparable to the results obtained by Pollen and Simon (2005) and Comino and Marengo (2010) for tree and shrub species. The C_r values estimated including very fine roots are much higher in comparison to the C_r values obtained after excluding these roots. However, the results of this study can still be utilized to make a selection of the most suitable species for shallow bank slope stabilization purposes.

4.4.5 Implications for revegetation in the Lower Colorado River Basin

The revegetation activities along the LCRB are mainly carried out within the geomorphic floodplain zone that includes toe, bank, and overbank zones. The increased stability provided by riparian vegetation is expected to reduce soil erosion and support shallow bank stabilization. All four shrub species studied in this work were found to have many fine roots in the topsoil, which is desirable for reducing concentrated flow erosion in the case of a spatially uniform distribution of root density in a plane (De Baets et al., 2007). In another work, the modeling results of two woody riparian species on critical conditions for streambank stability along the Upper Truckee River in California showed that the addition of vegetation has the same effect as reducing the angle of the bank face (Simon et al., 2006). The estimated root cohesion values in present study are higher than those reported by Simon et al. (2006) for riparian shrub species. However, the shrub species from the present study are expected to be effective only for shallow slope stabilization as their root distribution is most developed in the topsoil (0-0.3 m). A similar opinion was suggested by De Baets et al. (2008), who mentioned that Mediterranean shrub species might only be effective in stabilizing the top 0.5 m of hill slope or terrace walls.

The riparian vegetation in LCRB over the last few decades has been dominated by the invasive *Tamarix* spp., posing a challenge for resource managers to replace it with native vegetation. In a related study in a semi-arid region, an average root-reinforcement value of 2.5 kPa was reported for *Tamarix ramosissima* for the entire bank profile (Pollen-Bankhead et al., 2009). The removal of *Tamarix* from the bank, if not replaced by other vegetation, might cause bank instabilities along the LCR channel and its tributaries. The exact implication of *T. ramosissima* removal and revegetation with native riparian species will require a site specific modeling approach, which could allow quantification of the actual root reinforcement in a particular channel and tributary where bank stabilization is desired. Nevertheless, the estimated root cohesion values for four desert species in present study together with others could be useful for preliminary planning purposes.

4.5 Conclusions

This study shows that total RLD values of the studied plants in arid regions range between (7.27 km m^{-3} to 18.72 km m^{-3}), and these values are significantly higher for the topsoil (0-0.3 m) compared to the subsoil (>0.3 m), which can be attributed to the presence of many fine roots near the surface. Among the four species investigated, *A. lentiformis* and *L. andersonii* were found to have higher mean RAR compared to *A. occidentalis* and *L. tridentata*. The contribution of the fine roots in RAR was significantly higher than the contribution of the coarse roots at both soil depths (0-0.3 m and >0.3 m). Contrary to the lowest value of RD and RLD, *L. tridentata* showed higher root tensile strength followed by *L. andersonii*, *A. lentiformis*, and *A. occidentalis* in a decreasing order. It was found that, the maximum contribution to additional cohesion by *A. lentiformis* followed by *L. andersonii*, *L. tridentata*, and *A. occidentalis* in a decreasing order. The root cohesion values estimated by the Wu's model for all species were highest in the topsoil (0-0.3 m) and decreased with increasing soil depth. FBM model showed lower values of root cohesion, but followed the similar trend observed from Wu's model. The root cohesion values estimated from FBM model is less than those estimated by Wu's model by a reduction factor ranging between 0.35 to 0.56 for root diameter > 0.05 mm diameter.

The higher root cohesion associated with *A. lentiformis* and *L. andersonii* implies that these species are a good choice for revegetation purpose in order to strengthen the topsoil zone through root reinforcement. The native shrub species in the present study could contribute to the shallow bank slope stabilization, but the studied plants will not prevent mass movements occurring at greater depths, since at those depths roots occupation is too little to increase soil shear strength. To estimate the actual contribution of the additional root cohesion provided by these native shrub species, further studies are required field conditions. The observations made in this study may be useful for species selection in the framework of ongoing and future revegetation activities in the LCRB and similar riparian areas in the Southwest US.

CHAPTER 5

CONCLUSIONS

The research presented here consisted of three parts: 1) a removal of nutrients and metals by constructed and naturally created wetlands in the Las Vegas Valley (LVV); 2) a vegetation assessment for the nutrient uptake potential of macrophytes in arid and semi-arid wetlands; and 3) the estimation of root cohesion for desert shrub species in the riparian ecosystem of arid and semi-arid regions and its potential for streambank stabilization. The results generated from each study are related to each other and lead toward an understanding of wetland function for ecosystem and engineering services for water resource management in semi-arid environments.

The first study (Chapter 2) focused on understanding the potential of constructed and naturally created wetlands for pollutant removal in LVV wetlands. The hypothesis that wetlands perform as a pollutant sink, where vegetation is useful for pollutant removal on both constructed and naturally created wetlands is supported. The nutrient removal potential of a wetland system was dependent on plant biomass and density regardless of plant type. High TN (total nitrate) concentrations were measured in above-ground plant tissue but high TP (total phosphate) concentration was measured in below-ground plant tissue, which suggested that harvest of the root system would be necessary for maximum phosphorus removal. However, above-ground harvest would be sufficient for nitrogen removal from the natural and constructed wetlands in LVV. Overall, the results of this study suggested that different plant species have different nutrient uptake characteristics, mostly determined by the ambient nutrient and hydrologic conditions. Below-ground plant tissue exhibited high concentrations of arsenic and selenium among the four wetland sites tested. In comparison to cattail species, bulrush species were more effective at arsenic and selenium storage in below-ground plant tissue. The better performance of bulrush species could provide a clue for utilization of wetland vegetation in selenium removal. The findings of this study have important implications for better understanding ecological services for water quality improvements through constructed and naturally created wetlands.

In the second study (Chapter 3), the structural and functional attributes of wetlands were analyzed for the best ecosystem services from large scale wetlands such that the one in LVW. Vegetation mapping showed the dominance of *P. australis* and *T. domingensis* among all studied wetlands. The expansion of *P. australis* in the LVW had readily surpassed *T. domingensis*, established monogeneric stands, and was also associated with decreasing species richness. The change in structural attributes of vegetation, in turn, affected the ecosystem function by altering the nutrient biogeochemical cycling. The measured above-ground biomass (kg m^{-2}) of *T. domingensis* was higher than *P. australis* among all studied wetland sites. Despite high TN and TP concentration in *P. australis* plant tissue, a greater biomass accumulation potential of *T. domingensis* resulted in higher nutrient uptake per unit area. The hypothesis that larger the wetlands vegetation acreage the more efficient the ecosystem services are, despite vegetation types and climatic region is partially confirmed. The net above-ground standing stock of nutrients in the Las Vegas Wash wetlands was estimated for *P. australis* (approximately 26418.7 kg TN and 1264.1 kg TP) and for *T. domingensis* (approximately 5183.8 kg TN and 272.8 kg TP). The study concludes that in LVW, both *T. domingensis* and *P. australis* could be utilized for water quality improvement. It should be noted, however, plant uptake alone is not enough to improve water quality below regulatory thresholds from large scale wetlands, and managing dominant vegetation may be required for better nutrient removal efficiency. In this study, the nutrient uptake function of similar wetlands vegetation among different climatic regions were also compared, with the result that the short-term function of wetlands vegetation for biomass accumulation, plant tissue nutrient concentration, and standing stock was not significantly different among wetlands from humid and semi-arid regions.

The third study (Chapter 4) analyzed the engineering services provided by wetland vegetation for bank stabilization. The characteristics of root distributions, including root density, root length density, and root area ratio (RAR), were analyzed for four native riparian species. The root density and root length density among four shrub species were significantly higher for topsoil (0-0.5 m) compared to the subsoil (>0.5 m) for four species. Among the studied species, *A. lentiformis* and *L. andersonii* had higher mean RARs compared to *A. occidentalis* and *L. tridentata*. The contribution of the fine roots in RAR was significantly higher than the contribution of the coarse roots at both soil depths (0-0.3 m and >0.3 m). In situ tensile strength tests were performed on all species studied. Higher root tensile strength was measured for *L. tridentate* followed by *L. andersonii*, *A. lentiformis*, and *A. occidentalis* in decreasing order. Information from root distributions and tensile strength tests were used to estimate the additional root cohesion provided through desert shrub species. Root cohesion values were estimated using a simple perpendicular model (the Wu model) and a fiber bundle model (FBM). The root cohesion values estimated by the Wu model for all species tested were highest in the topsoil (0-0.3 m) and decreased with increasing soil depth. The hypothesis that the root permeated soils makes up a composite material that will enhance the soil strength is supported since all studied species were observed with measurable root cohesion values. The maximum root cohesion was estimated for *A. lentiformis* followed by *L. andersonii*, *L. tridentata*, and *A. occidentalis* in decreasing order. The FBM underestimated the root cohesion values from the Wu model by a reduction factor (0.35 to 0.56) for the four native species tested. The high root cohesion values for *A. lentiformis* and *L. andersonii* also implied that these species are beneficial to revegetation efforts. However, all four shrub species contributed only to shallow bank slope stabilization, but would not prevent mass movement at greater depths, since the root cohesion values significantly decreased with increased subsoil depth. The results from this study are useful for species selection purpose for the ongoing revegetation activities in the Lower Colorado River Basin and similar riparian areas in the Southwest US.

Some additional recommendations are included here for future studies:

The uptake mechanism of wetland vegetation had differential potential for nutrient and metal pollutants. Above-ground plant parts were more efficient for nutrient uptake, but metal uptake was significantly higher in the below-ground parts. These results should be considered cautiously, since the study period was relatively short. Therefore, long-term investigation considering treated wastewater hydraulic retention time is highly recommended for future research. Wetland microcosms with different nutrient composition, plant growth stage, and wastewater source, etc. should be investigated further and compared with field experiments.

Wetland vegetation can act as seasonal or longer-term storage of nutrients, and the resultant litter decomposition can result in remobilization of previously stored nutrients. Uptake of TN and TP has been shown to increase when plants are harvested annually during the peak growing season. A net release of nutrients often occurs in the fall and early spring as a result of decomposition and nutrient leaching of plant litter. Therefore, long-term monitoring and management studies are necessary to investigate the effects of species on seasonal patterns of nutrient uptake and release.

Recently, some wetland sites have become dominated by a *P. australis* monoculture. The interspecies competition between *T. domingensis* and *P. australis* should be investigated to elucidate the mechanisms of *P. australis* dominance. Experimental studies to identify improved growth conditions for native species in mixed-culture wetlands are recommended for future study.

The results obtained from the vegetation study are based on the short-term evaluation of nutrient uptake potential based on wetland structure and function. Understanding wetland function and processes over a longer period of time is particularly important in semi-arid and arid areas where surface water levels fluctuate both seasonally and inter-annually. Long-term studies are recommended to detail eutrophication processes in arid and semi-arid wetlands, which requires knowledge of vegetation decomposition and nutrient retention rates in these areas.

To estimate root cohesion, the present study considered relatively young plants of similar age, and the total number of species selected in the field study was relatively small. Different sizes and ages of plant species should be considered for further root distribution studies, and greater plant sampling frequencies are recommended for more definitive results. Root distribution can be influenced by number of environmental conditions therefore seasonal analysis is strongly recommended for future investigation.

Root tensile strength tests in the present study were performed using different instruments for root diameter (>0.5 mm) and very fine roots (<0.5 mm). The use of similar instrument is recommended for future tensile tests to achieve more consistent results. The tests, here, were performed at a reference site (i.e. Virgin River), but additional tensile strength tests should be performed under actual field conditions, and these values should be verified by applying direct soil shear tests for revegetation purposes.

APPENDIX A. WETLANDS DATABASE FOR NUTRIENT AND METAL IN PLANT TISSUE, SEDIMENT, AND WATER COLUMN

i) Plant tissue total phosphorous (TP) from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands.

Site	**Season**	**Plant Species**	**TP (%)**		
			Shoot	**Root**	**Total**
LVW	Winter	Typha	0.043	0.082	0.063
LVW	Winter	Typha	0.052	0.061	0.057
LVW	Winter	Typha	0.045	0.065	0.055
LVW	Winter	Typha	0.049	0.054	0.052
LVW	Winter	Typha	0.061	0.081	0.071
LVW	Winter	Typha	0.051	0.071	0.061
LVW	Winter	Typha	0.032	0.049	0.041
LVW	Winter	Typha	0.046	0.052	0.049
LVW	Winter	Typha	0.053	0.067	0.060
LVW	Winter	Typha	0.042	0.045	0.044
LVW	Winter	Typha	0.053	0.061	0.057
LVW	Winter	Typha	0.053	0.078	0.066
LVW	Winter	Typha	0.046	0.049	0.048
LVW	Winter	Typha	0.042	0.083	0.063
LVW	Spring	Typha	0.070	0.068	0.069
LVW	Spring	Typha	0.074	0.072	0.073
LVW	Spring	Typha	0.047	0.053	0.050
LVW	Spring	Typha	0.042	0.059	0.051
LVW	Spring	Typha	0.023	0.057	0.040
LVW	Spring	Typha	0.056	0.054	0.055
LVW	Spring	Typha	0.070	0.098	0.084
LVW	Spring	Typha	0.058	0.067	0.063
LVW	Spring	Typha	0.033	0.088	0.061
LVW	Spring	Typha	0.046	0.071	0.059
LVW	Spring	Typha	0.036	0.073	0.055
LVW	Spring	Typha	0.042	0.052	0.047
LVW	Spring	Typha	0.034	0.133	0.084
LVW	Spring	Typha	0.039	0.136	0.088
LVW	Spring	Typha	0.048	0.065	0.057
LVW	Spring	Typha	0.028	0.056	0.042
LVW	Summer	Typha	0.084	0.139	0.112
LVW	Summer	Typha	0.089	0.157	0.123
LVW	Summer	Typha	0.116	0.170	0.143

Site	**Season**	**Plant Species**	**TP (%)**		
			Shoot	**Root**	**Total**
LVW	Summer	Typha	0.103	0.148	0.126
LVW	Summer	Typha	0.073	0.145	0.109
LVW	Summer	Sch-cal	0.082	0.224	0.153
LVW	Summer	Sch-cal	0.082	0.259	0.171
HD	Winter	Sch-ac	0.048	0.087	0.068
HD	Winter	Sch-ac	0.044	0.083	0.064
HD	Winter	Sch-am	0.049	0.084	0.067
HD	Winter	Sch-am	0.063	0.076	0.070
HD	Winter	Sch-cal	0.061	0.089	0.075
HD	Winter	Sch-cal	0.069	0.079	0.074
HD	Winter	Sch-cal	0.086	0.088	0.087
HD	Winter	Sch-cal	0.066	0.076	0.071
HD	Spring	Sch-ac	0.169	0.205	0.187
HD	Spring	Sch-ac	0.120	0.170	0.145
HD	Spring	Sch-am	0.068	0.077	0.073
HD	Spring	Sch-am	0.064	0.650	0.357
HD	Spring	Sch-cal	0.060	0.171	0.116
HD	Spring	Sch-cal	0.056	0.019	0.038
HD	Spring	Sch-cal	0.069	0.126	0.098
HD	Spring	Sch-cal	0.075	0.123	0.099
HD	Spring	Sch-cal	0.086	0.153	0.120
HD	Spring	Sch-cal	0.084	0.169	0.127
HD	Spring	Typha	0.070	0.094	0.082
HD	Spring	Typha	0.060	0.097	0.079
HD	Spring	Typha	0.064	0.101	0.083
HD	Spring	Typha	0.071	0.105	0.088
HD	Spring	Typha	0.056	0.093	0.075
HD	Spring	Typha	0.052	0.091	0.072
HD	Summer	Typha	0.067	0.118	0.093
HD	Summer	Typha	0.082	0.138	0.110
HD	Summer	Typha	0.057	0.113	0.085
HD	Summer	Typha	0.091	0.146	0.119
HD	Summer	Sch-cal	0.188	0.225	0.207
HD	Summer	Sch-cal	0.229	0.251	0.240
HD	Summer	Sch-cal	0.143	0.179	0.161
HD	Summer	Sch-cal	0.155	0.186	0.171
PW	Winter	Sch-ac	0.015	0.020	0.018
PW	Winter	Sch-ac	0.017	0.022	0.020
PW	Winter	Sch-am	0.015	0.030	0.023
PW	Winter	Sch-am	0.013	0.016	0.015
PW	Winter	Sch-cal	0.022	0.024	0.023

Site	Season	Plant Species	TP (%)		
			Shoot	**Root**	**Total**
PW	Winter	Sch-cal	0.018	0.020	0.019
PW	Spring	Sch-ac	0.024	0.030	0.027
PW	Spring	Sch-ac	0.019	0.028	0.024
PW	Spring	Sch-am	0.016	0.027	0.022
PW	Spring	Sch-am	0.029	0.030	0.030
PW	Spring	Sch-cal	0.039	0.048	0.044
PW	Spring	Sch-cal	0.056	0.058	0.057
PW	Spring	Sch-cal	0.031	0.048	0.040
PW	Spring	Sch-cal	0.038	0.047	0.043
PW	Summer	Sch-am	0.006	0.017	0.012
PW	Summer	Sch-am	0.009	0.030	0.020
PW	Summer	Sch-am	0.021	0.017	0.019
PW	Summer	Sch-am	0.018	0.024	0.021
PW	Summer	Sch-am	0.103	0.078	0.091
PW	Summer	Sch-am	0.078	0.082	0.080
PW	Summer	Sch-am	0.039	0.089	0.064
PW	Summer	Sch-am	0.043	0.063	0.053
PW	Summer	Sch-am	0.087	0.047	0.067
PW	Summer	Sch-am	0.091	0.061	0.076
PW	Summer	Sch-am	0.087	0.059	0.073
PW	Summer	Sch-am	0.091	0.064	0.078
PW	Summer	Sch-am	0.050	0.097	0.074
PW	Summer	Sch-am	0.035	0.069	0.052
PW	Summer	Sch-am	0.039	0.047	0.043
PW	Summer	Sch-am	0.041	0.059	0.050
PW	Summer	Sch-am	0.043	0.054	0.049
PW	Summer	Sch-am	0.095	0.032	0.064
PW	Summer	Sch-am	0.041	0.060	0.051
PW	Summer	Sch-am	0.047	0.049	0.048
PW	Summer	Sch-am	0.077	0.068	0.073
PW	Summer	Sch-am	0.076	0.082	0.079
PW	Summer	Sch-cal	0.029	0.048	0.039
PW	Summer	Sch-cal	0.046	0.048	0.047
PW	Summer	Sch-cal	0.021	0.038	0.030
PW	Summer	Sch-cal	0.049	0.051	0.050
PW	Summer	Sch-cal	0.040	0.040	0.040
PW	Summer	Sch-cal	0.020	0.070	0.050
PW	Summer	Sch-cal	0.020	0.040	0.030
PW	Summer	Sch-cal	0.020	0.050	0.030
PW	Summer	Sch-cal	0.020	0.050	0.030
PW	Summer	Sch-cal	0.070	0.070	0.070

Site	Season	Plant Species	TP (%)		
			Shoot	Root	Total
PW	Summer	Sch-cal	0.070	0.060	0.060
PW	Summer	Sch-cal	0.030	0.050	0.040
PW	Summer	Sch-cal	0.020	0.060	0.040
PW	Summer	Sch-cal	0.030	0.070	0.050
PW	Summer	Sch-cal	0.030	0.080	0.060
PW	Summer	Sch-cal	0.030	0.040	0.040
PW	Summer	Sch-cal	0.030	0.070	0.050
PW	Summer	Typha	0.060	0.070	0.070
PW	Summer	Typha	0.070	0.050	0.060
PW	Summer	Typha	0.080	0.070	0.080
PW	Summer	Typha	0.060	0.050	0.060
PW	Summer	Typha	0.050	0.040	0.040
FW	Spring	Typha	0.020	0.040	0.030
FW	Spring	Typha	0.020	0.040	0.030
FW	Spring	Typha	0.020	0.020	0.020
FW	Spring	Typha	0.020	0.010	0.020
FW	Spring	Typha	0.020	0.020	0.020
FW	Spring	Typha	0.020	0.030	0.020
FW	Summer	Typha	0.050	0.100	0.070
FW	Summer	Typha	0.050	0.100	0.070
FW	Summer	Typha	0.060	0.080	0.070
FW	Summer	Typha	0.040	0.080	0.060
FW	Summer	Typha	0.050	0.110	0.080
FW	Summer	Typha	0.050	0.100	0.070
FW	Summer	Typha	0.050	0.080	0.060
FW	Summer	Typha	0.040	0.090	0.060
FW	Summer	Typha	0.070	0.080	0.080
FW	Summer	Typha	0.080	0.080	0.080

Note: Sch-cal: *Schoenoplectus californicus*, Sch-am: *Schoenoplectus americanus*, Sch-ac: *Schoenoplectus acutus* and Typha: *Typha domingensis*

ii) Water column total phosphorus (mg/L) from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands.

Site	Location	SNWA Location	Sampling Date	TP (mg/L)
				Water Column
LVW	Inlet	LW 6.85	Oct-07	0.160
LVW	Inlet	LW 6.85	Nov-07	0.120
LVW	Inlet	LW 6.85	Dec-07	0.120
LVW	Inlet	LW 6.85	Jan-07	0.094
LVW	Inlet	LW 6.85	Feb-07	0.130
LVW	Inlet	LW 6.85	Mar-07	0.093
LVW	Inlet	LW 6.85	Apr-07	0.160
LVW	Inlet	LW 6.85	Feb-08	0.084
LVW	Inlet	LW 6.85	Mar-08	0.080
LVW	Inlet	LW 6.85	Apr-08	0.130
LVW	Outlet	LW 5.9	Feb-07	0.100
LVW	Outlet	LW 5.9	Mar-07	0.150
LVW	Outlet	LW 5.9	Apr-07	0.130
LVW	Outlet	LW 5.9	Feb-08	0.140
LVW	Outlet	LW 5.9	Mar-08	0.140
LVW	Inlet	LW 5.9	May-07	0.150
LVW	Inlet	LW 5.9	Jun-07	0.130
LVW	Inlet	LW 5.9	Jul-07	0.120
LVW	Inlet	LW 5.9	Aug-07	0.110
LVW	Outlet	LW 5.9	May-07	0.150
LVW	Outlet	LW 5.9	Jun-07	0.140
LVW	Outlet	LW 5.9	Jul-07	0.130
LVW	Outlet	LW 5.9	Aug-07	0.120
HD	Inlet	HD1	Nov-08	1.610
HD	Inlet	HD1	Dec-08	0.840
HD	Inlet	HD1	Nov-07	0.600
HD	Inlet	HD1	Dec-07	0.920
HD	Outlet	HD4	Nov-08	1.120
HD	Outlet	HD4	Dec-08	0.810
HD	Outlet	HD4	Nov-07	0.950
HD	Outlet	HD4	Dec-07	0.850
HD	Inlet	HD1	Jan-07	1.020
HD	Inlet	HD1	Feb-07	2.010
HD	Inlet	HD1	Mar-07	0.940

Site	Location	SNWA Location	Sampling Date	TP (mg/L)
				Water Column
HD	Inlet	HD1	Apr-07	0.560
HD	Outlet	HD4	Jan-07	1.830
HD	Outlet	HD4	Feb-07	3.450
HD	Outlet	HD4	Mar-07	4.130
HD	Inlet	HD1	May-07	1.120
HD	Inlet	HD1	Jun-07	0.370
HD	Inlet	HD1	Jul-07	1.220
HD	Inlet	HD1	Aug-07	0.510
HD	Outlet	HD4	May-07	2.540
HD	Outlet	HD4	Jun-07	1.290
HD	Outlet	HD4	Jul-07	0.860
HD	Outlet	HD4	Aug-07	2.620
PW	Inlet	PW-Inlet	Nov-07	0.038
PW	Inlet	PW-Inlet	Dec-07	0.045
PW	Outlet	PW-Outlet	Nov-07	0.033
PW	Outlet	PW-Outlet	Dec-07	0.033
PW	Inlet	PW-Inlet	Jan-07	0.160
PW	Inlet	PW-Inlet	Feb-07	0.096
PW	Inlet	PW-Inlet	Mar-07	0.034
PW	Inlet	PW-Inlet	Jan-08	0.110
PW	Outlet	PW-Outlet	Jan-07	0.025
PW	Outlet	PW-Outlet	Feb-07	0.040
PW	Outlet	PW-Outlet	Mar-07	0.072
PW	Inlet	PW-Inlet	May-07	0.041
PW	Inlet	PW-Inlet	Jun-07	0.110
PW	Outlet	PW-Outlet	May-07	0.030
PW	Outlet	PW-Outlet	Jun-07	0.030
FW	Inlet	TW-DRI	Jan-07	0.070
FW	Inlet	TW-DRI	Feb-07	0.020
FW	Inlet	TW-DRI	Mar-07	0.030
FW	Inlet	TW-DIR	Jan-08	0.010
FW	Outlet	FW-0	Jan-07	0.020
FW	Outlet	FW-0	Feb-07	0.060
FW	Outlet	FW-0	Mar-07	0.030
FW	Outlet	FW-0	Jan-08	0.020
FW	Inlet	TW-DRI	May-07	0.130
FW	Inlet	TW-DRI	Jun-07	0.110
FW	Inlet	TW-DRI	Aug-07	0.050
FW	Inlet	TW-DRI	Jul-07	0.080

Site	Location	SNWA Location	Sampling Date	TP (mg/L)
				Water Column
FW	Outlet	FW-0	May-07	0.010
FW	Outlet	FW-0	Jun-07	0.020
FW	Outlet	FW-0	Jul-07	0.030
FW	Outlet	FW-0	Aug-07	0.050
FW	Inlet	TW-DRI	Nov-07	0.050
FW	Inlet	TW-DRI	Dec-07	0.060

iii) Sediment total phosphorus (TP) from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands

Site	Season	Location	TP (%)
			Sediment
LVW	Winter 08	Inlet	0.048
LVW	Winter 08	Inlet	0.045
LVW	Winter 08	Inlet	0.044
LVW	Winter 08	Inlet	0.042
LVW	Spring 09	Outlet	0.036
LVW	Spring 09	Outlet	0.038
LVW	Spring 09	Outlet	0.058
LVW	Spring 09	Outlet	0.056
LVW	Summer 09	Outlet	0.049
LVW	Summer 09	Outlet	0.047
LVW	Summer 09	Outlet	0.056
LVW	Summer 09	Inlet	0.057
HD	Winter 08	Inlet	1.610
HD	Winter 08	Inlet	0.840
HD	Winter 08	Inlet	0.600
HD	Winter 08	Inlet	0.920
HD	Winter 08	Inlet	1.120
HD	Winter 08	Inlet	0.810
HD	Winter 08	Outlet	0.950
HD	Winter 08	Outlet	0.850
HD	Spring 09	Outlet	1.020
HD	Spring 09	Outlet	2.010
HD	Spring 09	Outlet	0.940
HD	Spring 09	Inlet	0.560
HD	Spring 09	Inlet	1.830
HD	Spring 09	Inlet	3.450

Site	Season	Location	TP (%)
			Sediment
HD	Spring 09	Inlet	4.130
HD	Spring 09	Outlet	1.800
HD	Summer 09	Outlet	1.120
HD	Summer 09	Outlet	0.370
HD	Summer 09	Inlet	1.290
HD	Summer 09	Outlet	1.220
HD	Summer 09	Inlet	0.510
HD	Summer 09	Inlet	0.860
HD	Summer 09	Outlet	2.620
PW	Winter 08	Outlet	0.030
PW	Winter 08	Outlet	0.025
PW	Winter 08	Outlet	0.031
PW	Winter 08	Inlet	0.026
PW	Spring 09	Inlet	0.041
PW	Spring 09	Inlet	0.042
PW	Spring 09	Inlet	0.037
PW	Spring 09	Outlet	0.034
PW	Summer 09	Outlet	0.054
PW	Summer 09	Outlet	0.057
PW	Summer 09	Outlet	0.052
PW	Summer 09	Inlet	0.051
FW	Spring 09	Inlet	0.041
FW	Spring 09	Inlet	0.044
FW	Spring 09	Inlet	0.019
FW	Spring 09	Outlet	0.016
FW	Summer 09	Outlet	0.021
FW	Summer 09	Outlet	0.034
FW	Summer 09	Outlet	0.020
FW	Summer 09	Inlet	0.013
FW	Winter 08	Inlet	0.021
FW	Winter 08	Inlet	0.027
FW	Winter 08	Inlet	0.021
FW	Winter 08	Outlet	0.014

iv) Plant tissue total nitrogen (TN) from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands.

Site	Season	Plant	TN (%)		
			Shoot	Root	Total
LVW	Spring	Typha	1.77	1.20	1.48
LVW	Spring	Typha	1.68	1.25	1.46
LVW	Spring	Typha	1.81	1.12	1.46
LVW	Spring	Typha	1.71	1.60	1.65
LVW	Spring	Typha	1.40	1.12	1.26
LVW	Spring	Typha	1.40	1.05	1.22
LVW	Spring	Typha	0.94	0.84	0.89
LVW	Spring	Typha	0.86	0.73	0.79
LVW	Spring	Typha	1.90	1.30	1.60
LVW	Spring	Typha	1.88	1.35	1.61
LVW	Spring	Typha	1.88	1.14	1.51
LVW	Spring	Typha	1.69	1.21	1.45
LVW	Spring	Typha	1.69	1.24	1.46
LVW	Spring	Typha	0.89	0.51	0.70
LVW	Spring	Typha	0.96	1.15	1.05
LVW	Spring	Typha	1.56	1.95	1.75
LVW	Spring	Typha	1.83	1.44	1.63
LVW	Spring	Typha	2.74	1.03	1.88
LVW	Spring	Typha	2.68	1.04	1.86
LVW	Spring	Typha	2.05	1.26	1.65
LVW	Spring	Typha	1.56	1.42	1.49
LVW	Spring	Typha	0.83	1.69	1.26
LVW	Spring	Typha	0.92	2.12	1.52
LVW	Spring	Typha	1.34	1.00	1.17
LVW	Spring	Typha	2.91	1.52	2.21
LVW	Spring	Typha	1.68	1.03	1.35
LVW	Spring	Typha	1.18	0.86	1.02
LVW	Spring	Typha	1.95	0.89	1.42
LVW	Spring	Typha	1.76	0.94	1.35
LVW	Spring	Typha	2.32	1.46	1.89
LVW	Spring	Typha	1.90	1.07	1.48
LVW	Spring	Sch-cal	1.86	1.64	1.75
LVW	Spring	Sch-cal	1.84	1.50	1.67
LVW	Spring	Sch-cal	2.10	1.20	1.65

Site	Season	Plant	TN (%)		
			Shoot	Root	Total
LVW	Spring	Sch-cal	2.57	1.42	1.99
LVW	Spring	Sch-cal	2.62	1.34	1.98
LVW	Winter	Typha	1.58	1.10	1.34
LVW	Winter	Typha	1.18	0.82	1.00
LVW	Winter	Typha	0.94	0.84	0.89
LVW	Winter	Typha	0.86	0.79	0.82
LVW	Winter	Typha	1.39	1.46	1.42
LVW	Winter	Typha	0.95	0.78	0.86
LVW	Winter	Typha	2.32	1.07	1.69
LVW	Winter	Typha	0.97	0.78	0.88
LVW	Winter	Typha	1.05	1.29	1.17
LVW	Winter	Typha	1.36	0.86	1.11
LVW	Winter	Typha	1.92	1.12	1.52
LVW	Winter	Typha	2.32	1.26	1.79
LVW	Summer	Typha	2.23	1.55	1.89
LVW	Summer	Typha	1.96	1.48	1.72
LVW	Summer	Typha	1.79	1.04	1.41
LVW	Summer	Typha	2.17	1.01	1.59
LVW	Summer	Typha	1.82	1.80	1.81
LVW	Summer	Typha	1.73	1.76	1.74
LVW	Summer	Sch-cal	2.41	2.13	2.27
LVW	Summer	Sch-cal	2.51	2.15	2.33
LVW	Summer	Sch-cal	1.53	1.21	1.37
LVW	Summer	Sch-cal	1.70	1.42	1.56
HD	Spring	Sch-ac	1.01	1.06	1.03
HD	Spring	Sch-ac	1.26	1.06	1.16
HD	Spring	Sch-am	1.37	1.19	1.28
HD	Spring	Sch-am	1.43	1.24	1.33
HD	Spring	Sch-cal	1.68	1.36	1.52
HD	Spring	Sch-cal	1.70	1.35	1.52
HD	Spring	Sch-cal	0.80	1.30	1.05
HD	Spring	Sch-cal	0.97	1.27	1.12
HD	Spring	Sch-cal	1.41	1.47	1.44
HD	Spring	Sch-cal	1.42	1.58	1.50
HD	Spring	Typha	1.89	1.44	1.66
HD	Spring	Typha	1.77	1.70	1.73
HD	Spring	Typha	1.36	1.81	1.58

Site	**Season**	**Plant**	**TN (%)**		
			Shoot	**Root**	**Total**
HD	Spring	Typha	1.77	1.51	1.64
HD	Spring	Typha	1.79	1.51	1.65
HD	Winter	Sch-ac	1.15	1.15	1.15
HD	Winter	Sch-ac	1.40	1.43	1.42
HD	Winter	Sch-am	2.41	0.71	1.56
HD	Winter	Sch-am	1.26	1.12	1.19
HD	Winter	Sch-cal	1.26	1.40	1.33
HD	Winter	Sch-cal	1.58	1.05	1.31
HD	Winter	Sch-cal	1.25	2.41	1.83
HD	Winter	Sch-cal	1.30	0.85	1.07
HD	Summer	Typha	2.10	1.85	1.97
HD	Summer	Typha	2.10	1.90	2.00
HD	Summer	Typha	1.54	1.21	1.37
HD	Summer	Typha	1.32	1.27	1.29
HD	Summer	Sch-cal	2.09	1.41	1.75
HD	Summer	Sch-cal	2.21	2.02	2.11
HD	Summer	Sch-cal	2.33	1.52	1.92
HD	Summer	Sch-cal	2.46	1.25	1.85
PW	Summer	Typha	1.17	0.77	0.97
PW	Summer	Typha	0.63	0.91	0.77
PW	Summer	Sch-cal	0.58	1.27	0.92
PW	Summer	Sch-cal	1.46	1.25	1.35
PW	Summer	Sch-cal	1.57	1.28	1.43
PW	Summer	Sch-cal	1.29	0.86	1.07
PW	Summer	Sch-cal	1.18	0.89	1.04
PW	Summer	Sch-cal	1.04	1.06	1.05
PW	Summer	Sch-cal	1.22	0.90	1.06
PW	Summer	Sch-cal	0.92	1.02	0.97
PW	Summer	Sch-cal	0.79	0.47	0.63
PW	Summer	Sch-am	1.25	1.16	1.21
PW	Summer	Sch-am	1.58	1.25	1.42
PW	Summer	Sch-am	1.13	1.23	1.18
PW	Summer	Sch-am	1.38	0.72	1.05
PW	Summer	Sch-am	0.96	0.54	0.75
PW	Summer	Sch-am	1.90	1.42	1.66
PW	Summer	Sch-am	0.89	0.81	0.85
PW	Summer	Sch-am	0.94	0.73	0.84

Site	Season	Plant	TN (%)		
			Shoot	Root	Total
PW	Spring	Sch-ac	0.85	0.72	0.78
PW	Spring	Sch-ac	0.76	0.71	0.73
PW	Spring	Sch-am	0.91	0.79	0.85
PW	Spring	Sch-am	0.79	0.65	0.72
PW	Spring	Sch-cal	0.86	0.76	0.81
PW	Spring	Sch-cal	0.92	0.82	0.87
PW	Spring	Sch-cal	0.91	0.78	0.84
PW	Winter	Sch-ac	0.80	0.53	0.66
PW	Winter	Sch-am	0.73	0.63	0.68
PW	Winter	Sch-am	0.91	0.82	0.86
PW	Winter	Sch-am	0.63	0.52	0.57
PW	Winter	Sch-cal	0.82	0.75	0.78
PW	Winter	Sch-ac	0.68	0.61	0.64
FW	Spring	Typha	1.20	0.49	0.84
FW	Spring	Typha	1.15	0.58	0.86
FW	Spring	Typha	0.85	0.54	0.69
FW	Spring	Typha	0.50	0.45	0.47
FW	Spring	Typha	0.75	0.40	0.57
FW	Summer	Typha	1.24	0.81	1.02
FW	Summer	Typha	1.34	1.17	1.25
FW	Summer	Typha	1.56	1.05	1.30
FW	Summer	Typha	1.60	1.07	1.33
FW	Summer	Typha	1.16	0.73	0.94
FW	Summer	Typha	1.18	1.38	1.27
FW	Summer	Typha	1.33	0.74	1.03
FW	Summer	Typha	1.34	0.74	1.03
FW	Summer	Typha	1.12	0.81	0.96
FW	Summer	Typha	1.07	0.79	0.92
FW	Summer	Typha	1.53	0.91	1.21
FW	Summer	Typha	1.52	0.90	1.20
FW	Summer	Typha	1.67	0.99	1.32
FW	Summer	Typha	1.70	1.00	1.34
FW	Summer	Typha	1.56	0.99	1.27
FW	Summer	Typha	1.83	1.03	1.42
FW	Summer	Typha	1.30	1.10	1.19
FW	Summer	Typha	1.50	0.91	1.20

v) Water column total nitrogen (TN) from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands. (Ref: SNWA-database).

Site	Location	SNWA Location	Sampling Date	TN (mg/L)
				Water Column
LVW	Inlet	LW 6.85	Jan-07	14.0
LVW	Inlet	LW 6.85	Feb-07	15.0
LVW	Inlet	LW 6.85	Mar-07	16.0
LVW	Inlet	LW 6.85	Apr-07	13.0
LVW	Inlet	LW 6.85	Jan-07	14.0
LVW	Inlet	LW 6.85	Feb-07	15.0
LVW	Outlet	LW 5.9	Jan-07	14.0
LVW	Outlet	LW 5.9	Feb-07	14.0
LVW	Outlet	LW 5.9	Mar-07	14.0
LVW	Outlet	LW 5.9	Apr-07	14.0
LVW	Outlet	LW 5.9	Jan-07	14.0
LVW	Outlet	LW 5.9	Feb-07	15.0
LVW	Inlet	LW 6.85	Sep-07	16.0
LVW	Inlet	LW 6.85	Oct-07	14.0
LVW	Inlet	LW 6.85	Nov-07	17.0
LVW	Inlet	LW 6.85	Dec-07	14.0
LVW	Inlet	LW 6.85	Sep-07	17.0
LVW	Inlet	LW 6.85	Oct-07	15.0
LVW	Inlet	LW 6.85	Nov-07	14.0
LVW	Inlet	LW 6.85	Dec-07	16.0
LVW	Inlet	LW 6.85	Oct-07	13.0
LVW	Outlet	LW 5.9	Sep-07	14.0
LVW	Outlet	LW 5.9	Oct-07	15.0
LVW	Outlet	LW 5.9	Nov-07	16.0
LVW	Outlet	LW 5.9	Dec-07	13.0
LVW	Outlet	LW 5.9	Sep-07	14.0
LVW	Outlet	LW 5.9	Oct-07	15.0
LVW	Outlet	LW 5.9	Nov-07	14.0
LVW	Outlet	LW 5.9	Dec-07	14.0
LVW	Outlet	LW 5.9	Oct-07	15.0
LVW	Outlet	LW 5.9	Nov-07	14.0
LVW	Outlet	LW 5.9	Dec-07	15.0
LVW	Inlet	LW 6.85	May-07	13.0
LVW	Inlet	LW 6.85	Jun-07	16.0

Site	Location	SNWA Location	Sampling Date	TN (mg/L)
				Water Column
LVW	Inlet	LW 6.85	Jul-07	14.0
LVW	Inlet	LW 6.85	Aug-07	14.1
LVW	Inlet	LW 6.85	Jun-07	15.0
LVW	Inlet	LW 6.85	Jul-07	13.0
LVW	Inlet	LW 6.85	Aug-07	16.0
LVW	Outlet	LW 5.9	May-07	16.0
LVW	Outlet	LW 5.9	Jun-07	17.0
LVW	Outlet	LW 5.9	Jul-07	14.0
LVW	Outlet	LW 5.9	Aug-07	14.0
LVW	Outlet	LW 5.9	May-07	16.0
LVW	Outlet	LW 5.9	Jun-07	17.0
LVW	Outlet	LW 5.9	Jul-07	11.0
LVW	Outlet	LW 5.9	Aug-07	14.0
HD	Inlet	HD1	Jan-07	17.0
HD	Inlet	HD1	Feb-07	13.0
HD	Inlet	HD1	Mar-07	13.0
HD	Outlet	HD4	Jan-07	2.4
HD	Outlet	HD4	Feb-07	17.0
HD	Outlet	HD4	Mar-07	2.4
HD	Outlet	HD4	Apr-07	18.0
HD	Inlet	HD1	Sep-07	1.4
HD	Inlet	HD1	Oct-07	4.3
HD	Inlet	HD1	Nov-07	5.6
HD	Inlet	HD1	Dec-07	1.7
HD	Inlet	HD1	Nov-07	3.8
HD	Outlet	HD4	Oct-07	3.6
HD	Outlet	HD4	Nov-07	1.1
HD	Outlet	HD4	Dec-07	1.8
HD	Inlet	HD1	May-07	13.0
HD	Inlet	HD1	Jun-07	6.6
HD	Inlet	HD1	Jul-07	6.5
HD	Inlet	HD1	Aug-07	12.0
HD	Inlet	HD1	Jul-07	2.3
HD	Inlet	HD1	Aug-07	1.1
HD	Outlet	HD4	May-07	11.0
HD	Outlet	HD4	Jun-07	2.8
HD	Outlet	HD4	Jul-07	1.1
HD	Outlet	HD4	Aug-07	12.0

Site	Location	SNWA Location	Sampling Date	TN (mg/L)
				Water Column
HD	Outlet	HD4	Jul-07	2.1
HD	Outlet	HD4	Aug-07	1.1
PW	Inlet	PW-Inlet	May-07	8.1
PW	Inlet	PW-Inlet	Jul-07	10.0
PW	Inlet	PW-Inlet	Aug-07	8.5
PW	Outlet	PW-Outlet	May-07	8.1
PW	Outlet	PW-Outlet	Jun-07	10.0
PW	Outlet	PW-Outlet	Jul-07	9.5
PW	Outlet	PW-Outlet	Aug-07	8.5
PW	Outlet	PW-Outlet	May-07	7.8
PW	Outlet	PW-Outlet	Jun-07	10.0
PW	Outlet	PW-Outlet	Jul-07	9.6
PW	Outlet	PW-Outlet	Aug-07	8.3
PW	Inlet	PW-Inlet	Jan-08	8.8
PW	Inlet	PW-Inlet	Feb-08	9.6
PW	Inlet	PW-Inlet	Mar-08	10.0
PW	Outlet	PW-Outlet	Jan-08	8.9
PW	Outlet	PW-Outlet	Feb-08	9.6
PW	Outlet	PW-Outlet	Mar-08	9.0
PW	Outlet	PW-Outlet	Feb-07	8.8
PW	Outlet	PW-Outlet	Mar-07	9.5
PW	Outlet	PW-Outlet	Apr-07	10.0
PW	Inlet	PW-Inlet	Feb-07	9.9
PW	Inlet	PW-Inlet	Mar-07	9.1
PW	Inlet	PW-Inlet	Apr-07	8.9
PW	Outlet	PW-Outlet	Feb-07	9.5
PW	Outlet	PW-Outlet	Mar-07	8.9
PW	Outlet	PW-Outlet	Apr-07	8.8
PW	Outlet	PW-Outlet	Jan-08	7.5
PW	Outlet	PW-Outlet	Feb-08	8.1
PW	Outlet	PW-Outlet	Mar-08	9.6
PW	Outlet	PW-Outlet	Jan-08	8.8
PW	Outlet	PW-Outlet	Feb-08	8.7
PW	Inlet	PW-Inlet	Sep-07	8.1
PW	Inlet	PW-Inlet	Oct-07	8.8
PW	Outlet	PW-Outlet	Sep-07	9.0
PW	Outlet	PW-Outlet	Oct-07	8.7
PW	Outlet	PW-Outlet	Sep-07	8.9

Site	Location	SNWA Location	Sampling Date	TN (mg/L)
				Water Column
PW	Outlet	PW-Outlet	Oct-07	8.6
PW	Inlet	PW-Inlet	Aug-07	9.4
PW	Outlet	PW-Outlet	Sep-07	9.1
FW	Outlet	FW-0	Jan-07	4.3
FW	Outlet	FW-0	Feb-07	4.8
FW	Outlet	FW-0	Mar-07	4.5
FW	Outlet	FW-0	Jan-08	4.2
FW	Inlet	TW-DRI	Jan-07	2.1
FW	Inlet	TW-DRI	Feb-07	5.3
FW	Inlet	TW-DRI	Mar-07	3.5
FW	Inlet	TW-DRI	Feb-08	4.3
FW	Inlet	TW-DRI	Mar-08	4.3
FW	Inlet	TW-DRI	Jan-08	3.9
FW	Inlet	TW-DRI	Jan-07	5.2
FW	Inlet	TW-DRI	Feb-07	3.4
FW	Inlet	TW-DRI	Mar-07	5.1
FW	Inlet	TW-DRI	Jan-08	4.3
FW	Inlet	TW-DRI	Jan-07	3.1
FW	Outlet	FW-0	May-07	4.5
FW	Outlet	FW-0	Jun-07	3.6
FW	Outlet	FW-0	Jul-07	3.8
FW	Outlet	FW-0	Aug-07	3.5
FW	Outlet	FW-0	May-08	3.4
FW	Outlet	FW-0	Jun-08	4.1
FW	Outlet	FW-0	Jul-08	3.4
FW	Outlet	FW-0	Aug-08	3.6
FW	Inlet	TW-DRI	Oct-07	3.5
FW	Inlet	TW-DRI	Nov-07	5.4
FW	Inlet	TW-DRI	Dec-07	2.8
FW	Outlet	FW-0	Nov-07	4.3
FW	Outlet	FW-0	Dec-07	4.8

Note: Nearby sites were sampled for nutrients and metals in water column whenever insufficient samples were found in one location.

vi) Sediment total nitrogen (TN) from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands.

Site	**Season**	**Location**	**TN (%)**
			Sediment
LVW	Spring 09	Inlet	0.09
LVW	Spring 09	Inlet	0.07
LVW	Spring 09	Outlet	0.06
LVW	Spring 09	Outlet	0.08
LVW	Winter 08	Inlet	0.11
LVW	Winter 08	Inlet	0.15
LVW	Winter 08	Inlet	0.11
LVW	Winter 08	Outlet	0.13
LVW	Winter 08	Outlet	0.16
LVW	Winter 08	Outlet	0.14
LVW	Summer 08	Inlet	0.05
LVW	Summer 08	Inlet	0.07
LVW	Summer 08	Inlet	0.10
LVW	Summer 08	Outlet	0.06
LVW	Summer 08	Outlet	0.06
LVW	Summer 08	Outlet	0.06
HD	Spring 09	Inlet	0.05
HD	Spring 09	Inlet	0.05
HD	Spring 09	Outlet	0.05
HD	Spring 09	Outlet	0.06
HD	Winter 08	Inlet	0.07
HD	Winter 08	Inlet	0.07
HD	Winter 08	Inlet	0.06
HD	Winter 08	Outlet	0.04
HD	Winter 08	Outlet	0.04
HD	Winter 08	Outlet	0.04
HD	Summer 08	Inlet	0.08
HD	Summer 08	Inlet	0.07
HD	Summer 08	Inlet	0.09
HD	Summer 08	Outlet	0.07
HD	Summer 08	Outlet	0.05
HD	Summer 08	Outlet	0.06
PW	Summer 08	Inlet	0.05
PW	Summer 08	Inlet	0.05
PW	Summer 08	Inlet	0.04
PW	Summer 08	Outlet	0.05

Site	Season	Location	TN (%)
			Sediment
PW	Summer 08	Outlet	0.05
PW	Spring 09	Inlet	0.07
PW	Spring 09	Inlet	0.07
PW	Spring 09	Outlet	0.10
PW	Spring 09	Outlet	0.10
PW	Winter 08	Inlet	0.08
PW	Winter 08	Inlet	0.08
PW	Winter 08	Outlet	0.07
PW	Winter 08	Outlet	0.06
PW	Winter 08	Outlet	0.06
PW	Winter 08	Inlet	0.06
FW	Spring 09	Outlet	0.02
FW	Spring 09	Outlet	0.03
FW	Spring 09	Inlet	0.15
FW	Spring 09	Inlet	0.11
FW	Summer 08	Inlet	0.11
FW	Summer 08	Inlet	0.16
FW	Summer 08	Inlet	0.12
FW	Summer 08	Outlet	0.08
FW	Summer 08	Outlet	0.08
FW	Summer 08	Outlet	0.08
FW	Winter 08	Inlet	0.14
FW	Winter 08	Inlet	0.13
FW	Winter 08	Inlet	0.14
FW	Winter 08	Outlet	0.05
FW	Winter 08	Outlet	0.05
FW	Winter 08	Outlet	0.05

vii) Arsenic concentrations (As) in plant tissues from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands.

Site	**Season**	**Plant**	**Arsenic (µg/g)**		
			Shoot	**Root**	**Total**
LVW	Spring	Typha	5.21	1.32	3.26
LVW	Spring	Typha	4.23	2.12	3.17
LVW	Spring	Typha	5.31	2.86	4.08
LVW	Summer	Typha	10.10	1.39	5.74
LVW	Summer	Typha	9.86	1.16	5.51
LVW	Summer	Typha	9.16	1.12	5.14
LVW	Summer	Typha	3.60	1.53	2.56
LVW	Summer	Sch-cal	5.60	0.13	2.86
LVW	Summer	Sch-cal	3.02	1.44	2.23
LVW	Summer	Sch-cal	4.54	0.63	2.58
LVW	Winter	Typha	5.80	0.40	3.10
LVW	Winter	Typha	3.35	0.95	2.15
HD	Summer	Sch-cal	1.02	0.86	0.94
HD	Summer	Sch-cal	1.89	0.16	1.02
HD	Spring	Sch-ac	1.52	0.62	1.07
HD	Spring	Sch-cal	1.64	0.35	0.99
HD	Spring	Typha	2.06	0.25	1.19
HD	Winter	Sch-ac	2.05	0.84	1.44
HD	Winter	Sch-ac	2.00	0.70	1.35
HD	Winter	Sch-am	2.05	1.05	1.55
HD	Winter	Sch-am	2.56	0.40	1.48
HD	Winter	Sch-cal	2.51	0.25	1.37
HD	Winter	Sch-cal	1.72	0.35	1.02
PW	Winter	Sch-ac	8.05	1.60	4.80
PW	Winter	Sch-ac	6.24	1.51	3.87
PW	Winter	Sch-am	9.65	1.90	5.77
PW	Winter	Sch-am	8.41	2.60	5.50
PW	Winter	Sch-cal	10.20	1.25	5.72
PW	Summer	Sch-ac	10.21	3.90	7.05
PW	Summer	Sch-ac	8.34	2.30	5.32
PW	Summer	Sch-am	10.60	3.50	7.05
PW	Summer	Sch-am	9.41	4.88	7.14
PW	Summer	Sch-cal	13.91	0.20	7.05
PW	Summer	Sch-cal	12.28	0.35	6.31
PW	Summer	Sch-cal	5.85	1.20	3.52
PW	Summer	Sch-cal	6.85	3.54	5.19

Site	Season	Plant	Arsenic (µg/g)		
			Shoot	Root	Total
PW	Winter	Sch-cal	12.21	0.96	6.58
PW	Winter	Sch-cal	5.15	3.65	4.40
FW	Summer	Typha	3.50	0.60	2.05
FW	Summer	Typha	0.35	0.85	0.60
FW	Summer	Typha	2.21	0.56	1.38
FW	Summer	Typha	1.63	0.74	1.17

viii) Arsenic concentrations (As) from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands. (Ref: SNWA database)

Site	Location	Sampling Date	Arsenic (µg/L)
			Water Column
LVW	Inlet	Jan-07	5.9
LVW	Inlet	Feb-07	6.5
LVW	Inlet	Mar-07	7.2
LVW	Inlet	May-07	7.5
LVW	Inlet	Jun-07	6.2
LVW	Inlet	Jul-07	6.5
LVW	Inlet	Aug-07	5.7
LVW	Inlet	Sep-07	3.0
LVW	Inlet	Oct-07	1.8
LVW	Inlet	Nov-07	2.6
LVW	Inlet	Dec-07	5.1
LVW	Inlet	Sep-07	6.6
LVW	Inlet	Oct-07	6.6
LVW	Inlet	Nov-07	4.1
LVW	Inlet	Dec-07	7.3
LVW	Outlet	Jan-07	9.2
LVW	Outlet	Feb-07	9.8
LVW	Outlet	Mar-07	11.0
LVW	Outlet	May-07	9.4
LVW	Outlet	Jun-07	8.3
LVW	Outlet	Jul-07	8.4
LVW	Outlet	Aug-07	9.3
LVW	Outlet	Nov-07	8.9
LVW	Outlet	Dec-07	10.0

Site	Location	Sampling Date	Arsenic (µg/L)
			Water Column
LVW	Outlet	Oct-07	9.7
LVW	Outlet	Nov-07	7.2
LVW	Outlet	Dec-07	9.7
HD	Inlet	Jul-07	3.3
HD	Inlet	Aug-07	3.8
HD	Inlet	Nov-07	4.8
HD	Outlet	Feb-07	3.2
HD	Outlet	May-07	3.0
HD	Outlet	Jun-07	3.0
HD	Outlet	Jul-07	3.1
HD	Outlet	Aug-07	3.0
HD	Outlet	May-07	3.1
HD	Outlet	Jun-07	3.6
HD	Outlet	Jul-07	3.0
HD	Outlet	Aug-07	3.1
HD	Outlet	Sep-07	3.4
HD	Outlet	Nov-07	4.0
HD	Outlet	Dec-07	3.1
PW	Inlet	Feb-07	14.0
PW	Inlet	Mar-07	15.0
PW	Inlet	Apr-07	10.0
PW	Inlet	Feb-07	14.0
PW	Inlet	Mar-07	12.0
PW	Inlet	Apr-07	14.0
PW	Inlet	May-07	14.0
PW	Inlet	Jun-07	14.0
PW	Inlet	Jul-07	15.0
PW	Inlet	Aug-07	15.0
PW	Inlet	Sep-07	14.0
PW	Inlet	Oct-07	15.0
PW	Inlet	Nov-07	15.0
PW	Outlet	Feb-07	13.0
PW	Outlet	Mar-07	15.0
PW	Outlet	Apr-07	15.0
PW	Outlet	Feb-07	9.9
PW	Outlet	Mar-07	14.0
PW	Outlet	Apr-07	11.0
PW	Outlet	Feb-07	14.0

Site	Location	Sampling Date	Arsenic (µg/L)
			Water Column
PW	Outlet	Mar-07	14.0
PW	Outlet	Apr-07	12.0
PW	Outlet	Feb-07	15.0
PW	Outlet	Mar-07	13.0
PW	Outlet	Apr-07	15.0
PW	Outlet	Feb-07	15.0
PW	Outlet	Mar-07	9.8
PW	Outlet	Jun-07	14.0
PW	Outlet	Jul-07	16.0
PW	Outlet	Aug-07	15.0
PW	Outlet	May-07	14.0
PW	Outlet	Jun-07	14.0
PW	Outlet	Jul-07	15.0
PW	Outlet	Aug-07	15.0
PW	Outlet	Sep-07	15.0
PW	Outlet	Oct-07	15.0
PW	Outlet	Nov-07	15.0
FW	Outlet	Jan-01	6.4
FW	Outlet	Apr-01	7.5
FW	Outlet	Jan-02	8.1
FW	Outlet	Apr-02	7.2
FW	Outlet	Jan-03	5.2
FW	Outlet	Apr-03	4.8
FW	Outlet	Jan-04	7.4
FW	Outlet	Apr-04	5.4
FW	Outlet	Apr-05	7.0
FW	Outlet	Apr-06	5.2
FW	Outlet	Jan-07	4.1
FW	Outlet	Apr-07	4.9
FW	Outlet	Jan-08	4.5
FW	Outlet	Apr-08	4.5
FW	Outlet	Jul-01	6.2
FW	Outlet	Jul-02	9.2
FW	Outlet	Jul-03	5.8
FW	Outlet	Jul-04	5.1
FW	Outlet	Jul-05	8.5
FW	Outlet	Jul-06	5.5
FW	Outlet	Jul-07	5.5

Site	Location	Sampling Date	Arsenic (µg/L)
			Water Column
FW	Outlet	Oct-02	6.7
FW	Outlet	Oct-03	4.9
FW	Outlet	Oct-04	6.8
FW	Outlet	Oct-05	4.4
FW	Outlet	Oct-06	5.8
FW	Outlet	Oct-01	8.8

Note: Water quality data in LVW, HD & PW were selected for year 2007/08, for FW years 2001-2008, due to less frequent sampling.

ix) Arsenic concentrations (As) from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands.

Site	Location	Season	Arsenic (µg/g)
			Sediment
LVW	Inlet	Spring 09	3.50
LVW	Inlet	Spring 09	3.69
LVW	Inlet	Summer 09	3.86
LVW	Inlet	Summer 09	4.72
LVW	Inlet	Summer 09	4.71
LVW	Inlet	Summer 09	5.49
LVW	Inlet	Winter 08	5.68
LVW	Inlet	Winter 08	5.33
LVW	Outlet	Summer 09	3.63
LVW	Outlet	Summer 09	4.12
LVW	Outlet	Summer 09	5.78
LVW	Outlet	Summer 09	5.27
LVW	Outlet	Winter 08	4.72
LVW	Outlet	Winter 08	5.56
HD	Inlet	Spring 09	3.53
HD	Inlet	Spring 09	3.46
HD	Inlet	Summer 09	5.94
HD	Inlet	Summer 09	3.23
HD	Inlet	Summer 09	2.53
HD	Inlet	Summer 09	2.57
HD	Inlet	Winter 08	4.32
HD	Inlet	Winter 08	3.85

Site	Location	Season	Arsenic (µg/g)
			Sediment
HD	Outlet	Summer 09	2.74
HD	Outlet	Summer 09	3.05
HD	Outlet	Summer 09	3.38
HD	Outlet	Summer 09	3.64
HD	Outlet	Winter 08	2.32
HD	Outlet	Winter 08	2.52
PW	Inlet	Summer 09	5.99
PW	Inlet	Summer 09	6.21
PW	Inlet	Winter 08	6.35
PW	Outlet	Spring 09	5.61
PW	Outlet	Spring 09	4.03
PW	Outlet	Summer 09	6.81
PW	Outlet	Summer 09	6.90
PW	Outlet	Summer 09	4.11
PW	Outlet	Summer 09	3.63
PW	Outlet	Summer 09	5.80
PW	Outlet	Summer 09	8.30
PW	Outlet	Winter 08	6.38
PW	Outlet	Winter 08	7.25
FW	Inlet	Summer 09	2.44
FW	Inlet	Summer 09	3.02
FW	Inlet	Summer 09	1.99
FW	Inlet	Summer 09	2.45
FW	Inlet	Winter 08	3.03
FW	Inlet	Winter 08	3.56
FW	Outlet	Spring 09	3.86
FW	Outlet	Spring 09	2.51
FW	Outlet	Spring 09	3.06
FW	Outlet	Summer 09	4.37
FW	Outlet	Summer 09	3.89
FW	Outlet	Summer 09	2.38
FW	Outlet	Summer 09	2.53
FW	Outlet	Winter 08	3.61

x) Selenium concentrations (Se) in plant tissues from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands.

Site	**Season**	**Plant**	**Selenium (µg/g)**		
			Shoot	**Root**	**Total**
LVW	Spring	Typha	2.60	0.96	1.78
LVW	Spring	Typha	1.36	0.87	1.11
LVW	Spring	Typha	1.70	1.02	1.36
LVW	Spring	Typha	3.62	0.58	2.10
LVW	Summer	Typha	1.80	1.34	1.57
LVW	Summer	Typha	1.82	0.72	1.27
LVW	Summer	Typha	2.20	0.76	1.48
LVW	Summer	Typha	1.54	0.67	1.10
LVW	Summer	Sch-cal	1.58	0.58	1.08
LVW	Summer	Sch-cal	1.32	0.69	1.00
LVW	Winter	Typha	8.30	4.45	6.37
LVW	Winter	Typha	14.35	2.95	8.65
HD	Summer	Sch-cal	2.48	0.64	1.56
HD	Summer	Sch-cal	2.10	0.72	1.41
HD	Spring	Sch-am	1.80	1.80	1.80
HD	Spring	Sch-ac	4.46	1.80	3.13
HD	Spring	Sch-cal	2.90	1.50	2.20
HD	Spring	Typha	1.62	0.72	1.17
HD	Spring	Typha	2.38	0.59	1.48
HD	Winter	Sch-ac	6.40	2.45	4.42
HD	Winter	Sch-ac	6.85	3.50	5.17
HD	Winter	Sch-am	6.45	1.00	3.72
HD	Winter	Sch-am	5.45	2.15	3.80
HD	Winter	Sch-cal	1.95	0.50	1.22
HD	Winter	Sch-cal	1.40	2.40	1.90
PW	Winter	Sch-ac	11.65	9.30	10.47
PW	Winter	Sch-ac	14.80	8.70	11.75
PW	Winter	Sch-am	20.15	11.45	15.80
PW	Winter	Sch-am	18.11	14.53	16.32
PW	Winter	Sch-cal	8.60	6.45	7.52
PW	Summer	Sch-ac	12.00	2.45	7.22
PW	Summer	Sch-ac	9.65	3.27	6.46
PW	Summer	Sch-am	21.75	11.75	16.75
PW	Summer	Sch-am	17.56	15.28	16.42
PW	Summer	Sch-cal	5.90	2.20	4.05
PW	Summer	Sch-cal	7.54	5.21	6.37

Site	Season	Plant	Selenium (µg/g)		
			Shoot	Root	Total
PW	Summer	Sch-cal	8.00	4.15	6.07
PW	Summer	Sch-cal	6.38	0.00	6.38
PW	Winter	Sch-cal	5.20	4.21	4.70
PW	Winter	Sch-cal	14.70	3.65	9.17
PW	Winter	Sch-cal	17.51	5.36	11.43
FW	Summer	Typha	2.45	1.20	1.82
FW	Summer	Typha	1.90	0.15	1.02
FW	Summer	Typha	1.32	0.82	1.07
FW	Summer	Typha	1.76	0.72	1.24

Appendix xi) Selenium concentrations (Se) in Water Column from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands. (Ref: SNWA database).

Site	Location	Sampling Date	Selenium (µg/L)
			Water Column
LVW	Inlet	Jan-07	2.6
LVW	Inlet	Feb-07	2.9
LVW	Inlet	Mar-07	2.8
LVW	Inlet	Jan-07	3.8
LVW	Inlet	Feb-07	4.1
LVW	Inlet	Mar-07	3.9
LVW	Inlet	May-07	2.7
LVW	Inlet	Jun-07	2.7
LVW	Inlet	Jul-07	2.9
LVW	Inlet	Aug-07	2.6
LVW	Inlet	May-07	3.7
LVW	Inlet	Jun-07	3.3
LVW	Inlet	Jul-07	3.6
LVW	Inlet	Aug-07	3.3
LVW	Inlet	Sep-07	3.0
LVW	Inlet	Oct-07	2.6
LVW	Inlet	Nov-07	2.9
LVW	Inlet	Dec-07	2.7
LVW	Inlet	Sep-07	3.6
LVW	Inlet	Oct-07	4.0
LVW	Inlet	Nov-07	3.9

Site	Location	Sampling Date	Selenium (µg/L)
			Water Column
LVW	Inlet	Dec-07	3.7
LVW	Inlet	Nov-07	3.1
LVW	Inlet	Dec-07	4.2
LVW	Outlet	Jan-07	3.4
LVW	Outlet	Feb-07	3.7
LVW	Outlet	Mar-07	3.6
LVW	Outlet	May-07	3.2
LVW	Outlet	Jun-07	2.8
LVW	Outlet	Jul-07	2.9
LVW	Outlet	Aug-07	3.1
LVW	Outlet	Oct-07	4.0
LVW	Outlet	Nov-07	3.4
LVW	Outlet	Dec-07	3.1
LVW	Outlet	Nov-07	3.2
LVW	Outlet	Dec-07	3.3
HD	Inlet	Jan-07	1.6
HD	Inlet	Feb-07	2.6
HD	Inlet	Jul-07	2.1
HD	Inlet	Aug-07	2.2
HD	Inlet	Dec-07	2.0
HD	Outlet	May-07	2.0
HD	Outlet	Jun-07	1.9
HD	Outlet	Jul-07	1.6
HD	Outlet	Aug-07	1.8
HD	Outlet	May-07	2.0
HD	Outlet	Jun-07	1.3
HD	Outlet	Jul-07	1.2
HD	Outlet	Aug-07	2.1
PW	Inlet	Jan-07	9.3
PW	Inlet	Feb-07	11.0
PW	Inlet	Mar-07	9.8
PW	Inlet	Jan-07	11.0
PW	Inlet	Feb-07	12.0
PW	Inlet	Mar-07	11.0
PW	Inlet	May-07	10.0
PW	Inlet	Jun-07	11.0
PW	Inlet	Jul-07	11.0
PW	Inlet	Aug-07	10.0

Site	**Location**	**Sampling Date**	**Selenium (µg/L)**
			Water Column
PW	Inlet	Oct-07	9.8
PW	Inlet	Nov-07	11.0
PW	Inlet	Dec-07	11.0
PW	Outlet	Jan-07	9.1
PW	Outlet	Feb-07	10.0
PW	Outlet	Mar-07	12.0
PW	Outlet	Jan-07	11.0
PW	Outlet	Feb-07	11.0
PW	Outlet	Mar-07	11.0
PW	Outlet	Jan-07	11.0
PW	Outlet	Feb-07	11.0
PW	Outlet	Mar-07	12.0
PW	Outlet	Jan-07	12.0
PW	Outlet	Feb-07	8.6
PW	Outlet	Mar-07	9.2
PW	Outlet	Jan-08	11.0
PW	Outlet	Feb-08	10.0
PW	Outlet	May-07	10.0
PW	Outlet	Jul-07	11.0
PW	Outlet	Aug-07	11.0
PW	Outlet	May-07	9.9
PW	Outlet	Jun-07	11.0
PW	Outlet	Jul-07	11.0
PW	Outlet	Aug-07	11.0
PW	Outlet	Oct-07	11.0
PW	Outlet	Nov-07	11.0
PW	Outlet	Dec-07	11.0
FW	Inlet	Jan-07	8.6
FW	Inlet	Feb-07	8.4

Appendix xii) Selenium concentrations (Se) in sediment from the Las Vegas Wash (LVW), Flamingo Wash (FW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD) and Pittman Wash Pilot Wetlands (PW) wetlands.

Site	**Location**	**Season**	**Selenium (µg/g)**
			Sediment
LVMW	Inlet	Spring 09	3.50
LVMW	Inlet	Spring 09	3.69
LVMW	Inlet	Summer 09	3.86
LVMW	Inlet	Summer 09	4.72
LVMW	Inlet	Summer 09	4.71
LVMW	Inlet	Summer 09	5.49
LVMW	Inlet	Winter 08	5.68
LVMW	Inlet	Winter 08	5.33
LVMW	Outlet	Summer 09	3.63
LVMW	Outlet	Summer 09	4.12
LVMW	Outlet	Summer 09	5.78
LVMW	Outlet	Summer 09	5.27
LVMW	Outlet	Winter 08	4.72
LVMW	Outlet	Winter 08	5.56
HD	Inlet	Spring 09	3.53
HD	Inlet	Spring 09	3.46
HD	Inlet	Summer 09	5.94
HD	Inlet	Summer 09	3.23
HD	Inlet	Summer 09	2.53
HD	Inlet	Summer 09	2.57
HD	Inlet	Winter 08	4.32
HD	Inlet	Winter 08	3.85
HD	Outlet	Summer 09	2.74
HD	Outlet	Summer 09	3.38
HD	Outlet	Summer 09	3.64
HD	Outlet	Winter 08	2.32
HD	Outlet	Winter 08	2.52
PW	Inlet	Summer 09	5.99
PW	Inlet	Summer 09	6.21
PW	Inlet	Winter 08	7.60
PW	Inlet	Winter 08	6.35
PW	Outlet	Spring 09	5.60
PW	Outlet	Spring 09	4.03
PW	Outlet	Summer 09	6.80
PW	Outlet	Summer 09	6.90

Site	**Location**	**Season**	**Selenium (µg/g)**
			Sediment
PW	Outlet	Summer 09	4.11
PW	Outlet	Summer 09	3.63
PW	Outlet	Summer 09	5.80
PW	Outlet	Summer 09	8.30
PW	Outlet	Winter 08	6.38
PW	Outlet	Winter 08	7.25
FW	Inlet	Summer 09	2.44
FW	Inlet	Summer 09	3.02
FW	Inlet	Summer 09	1.99
FW	Inlet	Summer 09	2.45
FW	Inlet	Winter 08	3.03
FW	Inlet	Winter 08	3.56
FW	Outlet	Spring 09	3.86
FW	Outlet	Spring 09	2.51
FW	Outlet	Spring 09	3.06
FW	Outlet	Summer 09	4.37
FW	Outlet	Summer 09	3.89
FW	Outlet	Summer 09	2.38
FW	Outlet	Summer 09	2.53
FW	Outlet	Winter 08	3.61

xiii) Plant tissue total phosphorous (TP) and total nitrogen (TN) from the Las Vegas Wash (LVW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD), Pittman Wash Pilot Wetlands (PW) and Flamingo Wash (FW)

Site	**Season**	**Plant**	**TP (%)**		
			Shoot	**Root**	**Total**
LVW	Fall	Typha	0.07	0.12	0.09
LVW	Fall	Typha	0.18	0.15	0.17
LVW	Fall	Typha	0.07	0.09	0.08
LVW	Fall	Typha	0.09	0.07	0.08
HD	Fall	Sch-cal	0.05	0.08	0.06
HD	Fall	Sch-cal	0.08	0.07	0.07
HD	Fall	Sch-cal	0.10	0.16	0.13
HD	Fall	Sch-cal	0.07	0.14	0.10
PW	Fall	Sch-cal	0.05	0.09	0.07
PW	Fall	Sch-cal	0.01	0.04	0.03
PW	Fall	Sch-cal	0.02	0.03	0.03
PW	Fall	Sch-cal	0.02	0.05	0.04
FW	Fall	Typha	0.02	0.08	0.05
FW	Fall	Typha	0.02	0.04	0.03
FW	Fall	Typha	0.02	0.05	0.03
FW	Fall	Typha	0.02	0.04	0.03
Site	**Season**	**Plant**	**TN (%)**		
			Shoot	**Root**	**Total**
LVW	Fall	Typha	2.28	1.12	1.70
LVW	Fall	Typha	3.18	1.05	2.12
LVW	Fall	Typha	0.75	0.84	0.80
LVW	Fall	Typha	0.96	0.73	0.85
HD	Fall	Sch-cal	1.46	1.58	1.52
HD	Fall	Sch-cal	1.24	1.09	1.17
HD	Fall	Sch-cal	2.88	1.04	1.96
HD	Fall	Sch-cal	1.69	1.31	1.50
PW	Fall	Sch-cal	2.58	0.68	1.63
PW	Fall	Sch-cal	2.08	0.91	1.50
PW	Fall	Sch-cal	1.28	0.78	1.03
PW	Fall	Sch-cal	2.13	0.95	1.54
FW	Fall	Typha	1.24	0.72	0.98
FW	Fall	Typha	1.18	0.87	1.03
FW	Fall	Typha	1.15	0.63	0.89
FW	Fall	Typha	1.35	0.83	1.09

xiv) Annual average seasonal variation for plant tissue nutrients (TN, TP) and metals (As, Se) plant tissue nutrient concentration from the Las Vegas Wash (LVW), Demonstration Wetland at the City of Henderson Water Reclamation Facility (HD), Pittman Wash Pilot Wetlands (PW) and Flamingo Wash (FW)

Wetland sites	TP %		
	Winter	Spring	Summer
LVW	0.06±0.002	0.06±0.003	0.14±0.007
HD	0.07±0.002	0.11±0.01	0.15±0.01
PW	0.02±0.001	0.04±0.004	0.05±0.002
FW		0.03±0.002	0.08±0.002
Wetland sites	TN %		
	Winter	Spring	Summer
LVW	1.21±0.09	1.49±0.05	1.77±0.01
HD	1.36±0.08	1.40±0.06	1.79±0.10
PW	0.70±0.04	0.80±0.02	1.07±0.04
FW		0.69±0.07	1.19±0.03
Wetland sites	As (µg/g)		
	Winter	Spring	Summer
LVW	2.63±0.47	3.51±0.28	3.81±0.59
HD	1.37±0.07	1.09±0.05	0.98±0.04
PW	5.17±0.33		6.08±0.46
FW			1.30±0.29
Wetland sites	Se (µg/g)		
	Winter	Spring	Summer
LVW	7.51±1.13	1.59±0.21	1.27±0.08
HD	3.38±0.61	1.96±0.33	1.49±0.07
PW	10.9±1.04		8.72±1.74
FW			1.29±0.18

APPENDIX B. ROOT DIAMETER AND ROOT TENSILE STRENGTH OF NATIVE XEORIPARIAN SPECIES

Artiplex lentiformis		***Lycium andersonii***	
Average Diameter (mm)	**Root Tensile Tr (Mpa)**	**Average Diameter (mm)**	**Root Tensile Tr (Mpa)**
5.20	4.92	1.50	16.94
1.50	21.15	1.00	26.72
1.30	41.39	0.90	54.85
2.90	12.95	0.70	59.69
1.10	49.49	0.80	45.70
1.0	45.03	5.50	12.58
6.30	14.81	0.60	77.73
6.40	7.82	3.00	16.88
4.30	8.21	0.70	59.69
3.90	11.45	3.60	17.39
3.90	5.90	5.00	10.68
4.30	4.42	3.10	19.24
4.80	5.11	0.60	84.75
3.90	1.97	0.40	63.66
2.90	18.71	0.30	49.51
4.00	20.69	0.50	50.93
2.90	21.36	0.50	50.93
6.50	4.53	0.20	127.32
2.30	27.78	0.30	91.96
2.70	20.39	0.30	84.88
0.65	96.22	0.40	59.68
0.50	106.93	0.20	143.24
2.65	4.89	0.20	127.32
1.60	9.45	0.40	31.83
2.35	17.67	0.40	31.83
3.70	5.37	1.40	27.19
2.05	11.48	1.00	25.43
2.90	12.66	1.10	29.39
1.10	68.12	1.70	14.05
1.15	39.35	2.20	12.31
1.30	32.29	2.40	11.88
0.20	79.58	3.40	16.85
0.20	95.49	1.90	11.61
0.20	159.15	1.80	11.36
0.20	111.41	1.80	11.36

Artiplex lentiformis		*Lycium andersonii*	
Average Diameter (mm)	**Root Tensile Tr (Mpa)**	**Average Diameter (mm)**	**Root Tensile Tr (Mpa)**
0.25	112.04	0.50	30.56
0.35	62.36	0.40	47.75
0.25	122.23	0.80	10.94
0.20	95.49	0.60	35.37
0.20	143.24	0.80	13.93
0.25	122.23	0.70	23.39
0.20	95.49	0.60	34.48
0.30	70.74	0.50	53.48
0.30	87.00	0.30	159.15
0.20	143.24	0.30	173.30
0.35	54.57	0.30	187.45
		0.30	201.60
		0.80	30.34
		0.60	57.47
		0.80	34.32
		0.60	64.55
		0.80	38.30

Larrea tridentata		*Allenrolfea occidentalis*	
Average Diameter (mm)	**Root Tensile Tr (Mpa)**	**Average Diameter (mm)**	**Root Tensile Tr (Mpa)**
1.00	35.57	2.60	12.71
1.00	27.97	1.90	17.43
1.10	36.73	1.10	29.97
0.70	46.75	0.90	29.28
1.30	24.80	1.30	21.99
1.40	25.26	1.80	21.13
1.00	35.57	1.20	38.76
1.00	38.10	0.90	40.90
0.80	43.72	1.20	31.70
1.30	39.78	0.70	33.49
1.10	27.31	0.70	40.90
0.80	59.54	2.20	19.88
1.00	27.98	1.30	23.67
0.70	72.60	0.90	34.62
2.30	22.27	1.10	29.76
1.50	16.93	0.80	30.85
1.10	45.08	1.00	14.49
1.30	39.75	1.50	15.32
0.90	76.72	1.30	34.50
0.60	98.86	1.20	40.49
0.80	39.80	0.80	33.00
0.80	97.10	0.60	44.23
0.90	84.56	0.70	51.97
1.70	24.58	0.30	49.51
0.90	81.41	0.40	43.77
0.70	95.85	0.20	95.49
0.60	95.34	0.50	40.74
0.60	98.86	0.40	31.83
0.60	98.86	0.40	39.79
1.60	32.66	0.40	55.70
0.30	106.10	0.30	63.66
0.20	119.37	0.20	79.58
0.40	67.64	0.20	79.58
0.30	99.03	0.40	59.68
0.30	95.49	0.40	39.79

Larrea tridentata		*Allenrolfea occidentalis*	
Average Diameter (mm)	**Root Tensile Tr (Mpa)**	**Average Diameter (mm)**	**Root Tensile Tr (Mpa)**
0.30	106.10	3.80	9.56
0.30	91.96	1.30	23.29
0.50	45.84	1.80	20.75
0.20	111.41	1.00	29.29
0.30	49.51	1.10	25.37
0.40	59.68	2.00	19.96
0.50	48.38	2.20	22.78
0.30	84.88	2.10	15.24
		1.10	29.40
		1.50	25.37
		1.10	35.67
		2.50	18.04
		2.10	22.69
		3.60	8.50
		2.50	17.03
		0.50	45.84
		0.50	45.84
		0.40	55.70
		0.50	50.93

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