DETRITUS PROCESSING AND ECOLOGICAL STRUCTURE OF CONSTRUCTED WETPONDS WITH CONTRASTING AGE AND WATERSHED CHARACTERISTICS

THESIS

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INTRODUCTION

Wetlands are recognized for their ability to act as pollution filter systems, and serve many other roles in watersheds (EPA 1985, EPA 1987, Kent 2001). Only since the 1970's have researchers and agencies in the United States considered the use of wetlands in the treatment of wastewater (EPA 1987, Mitsch and Gosselink 2000), and used them to cleanse polluted waters (EPA 1987). However, concerns over using wetlands to treat wastewater include the possible harmful effects of toxic materials and pathogens in wastewaters, the long-term degradation of wetlands due to the additional nutrient and hydraulic loading from wastewater discharges, and whether or not natural wetlands can treat a continuous inflow of wastewater (Wetzel 1993, Hammer 1996, Mitsch and Gosselink 2000). Modification of naturally occurring wetlands for treatment purposes now requires approval under Section 404 of the Clean Water Act, and review under the National Environmental Protection Act (EPA 1987).

Historically, wetlands were regarded as wasted land to be filled in and developed. As a result, only 47% (42 million ha) of historical United States natural wetlands still exist (Mitsch and Gosselink 2000). Under the protection of the Clean Water Act, wetlands today are recognized for their multiple benefits. However, because of the potential harmful effects associated with using natural wetlands to treat wastewater, interest has increased in the use of constructed wetlands and constructed wetponds for wastewater treatment (EPA 1987, Davies and Bavor 2000). The EPA (1987) defines constructed wetlands as "...engineered systems that have been designed and constructed to employ wetland type vegetation to assist treating wastewater in a more controlled environment than occurs in natural wetlands." Hammer (1996) defines them as "...consist[ing] of former terrestrial environments that have been modified to create poorly drained soils, wetland flora and fauna for the primary purpose of contaminant or pollutant removal from wastewater." Constructed wetlands can be built at almost any location (EPA 1987), and used to treat municipal wastewater, mine drainage waste, runoff produced from residential and commercial developments, agricultural sources, parking lots, roadways and highways (Mitsch and Gosselink 2000). They are shallow systems that fill and drain, are densely covered with emergent plants (Wong et al. 1999), and have either standing water (surface-flow) or have a porous bottom through which wastewater passes to the outlet point of the wetland (subsurface-flow constructed wetlands, Mitsch and Gosselink 2000).

Constructed wetponds, wet detention basins (Flatt and Solanki 1995) or water pollution control ponds (Davies and Bavor 2000) are different than constructed wetlands because they are deeper, have a smaller range of water level fluctuation, and wetland vegetation is planted around the perimeter of the wetpond (Wong et al. 1999). The function of a wetpond is to capture runoff and allow settling of solids and particulate, and involves the integration of engineering and ecological processes. By maintaining a permanent pool, constructed wetponds remove pollutants through enhanced particle settling and biological activities before discharging the treated runoff into a waterway (Flatt and Solanki 1995).

Although variable, constructed wetpond design is based on three criteria: hydraulic residence time, a vegetative shelf, and wetpond design structure. Hydraulic residence time refers to the length of time that stormwater is held within the detention basin. Wetponds are designed to hold stormwater for a designated time period to allow sedimentation and for biological treatment to occur before it is flushed by additional rain events (Rushton et al. 1995). The longer the hydraulic residence time the greater the treatment effect (Zarriello 1990, Glick and Chang 1998). A minimum of 14 days hydraulic residence time is considered adequate (Schueler 1987, Rushton et al. 1995b), since shorter times result in a lower nutrient and sediment removal rate (Rushton et al. 1995b, Rushton et al. 1997, Mitsch and Gosselink 2000).

A vegetative shelf provides a littoral zone to support growth of emergent wetland vegetation. Vegetation decreases the velocity of stormwater runoff (Hammer 1996), allows time for sediment to settle, and shades and moderates water temperature (DeBusk and DeBusk 2001). Presence of plants in the wetpond aids in biological uptake of nutrients associated with stormwater runoff (Flatt and Solanki 1995, DeBusk et al. 1996, Davies and Bavor 2000). Wetpond plant selection should preferably be those native to the area that are capable of removing pollutants (Reddy and DeBusk 1985, Bachand and Horne 2000a, DeBusk and DeBusk 2001), and should avoid invasive or undesirable species. Cattail (*Typha spp.*) and duckweed (*Lemna minor*) show the ability to uptake lead and cadmium (Debusk et al. 1996); unfortunately, these species are also highly invasive (Niering 1990). Water hyacinth (*Eichhornia crassipes*), also a highly invasive plant, pennywort (*Hydrocotyle umbellata*) and duckweeds (*Lemna, Spirodella, and Wolffia sp.*) are used in macrophyte-based wastewater systems for their nutrient uptake abilities (Brix 1993). Planting a wetpond at its conception eliminates or retards the establishment of invasive or undesirable plants (Reinartz and Warne 1993, Hammer 1996, Glick and Chang 1998).

The design of a wetpond includes a permanent pool, extended detention pool and an optional flood detention pool (COA 1997). A permanent pool is designed to capture and retain the entire quantity of runoff from a watershed that would be produced during a two-week period of the wettest month of the year. This portion of the wetpond always contains water (Glick and Chang 1998). It enhances particle settling, decay processes and biological uptake, and the removal of particulate and dissolved pollutants (Flatt and Solanki 1995). The extended detention pool is designed to hold water from a two-year rain event and sits above the permanent pool elevation. The extended detention pool reduces turbulence in the wetpond and increases the time for sedimentation to occur by reducing the pond flow-through rate (COA 1997). The flood detention pool is designed to contain the 2-, 10-, 25-, and 100-year rain event (Glick and Chang 1998). Wetpond depth is also an important component, as shallower wetponds generally have higher dissolved oxygen concentrations, providing better pollutant removal efficiencies and more desirable aquatic habitat (Rushton et al. 1995b).

To determine effectiveness of wetponds and wetlands designed for the treatment of wastewater, structural characteristics such as engineering and design (Rushton et al. 1995b, Glick and Chang 1998), water quality and hydrology (Rushton and Dye 1990, Zarriello 1990, Crites et al. 1997, Walker 1998, Frankenback et al. 1999, Steer et al. 2002), the role of vegetation in nitrate removal rates (Bachand and Horne 2000b), sediment sampling (Walker and Hurl 2002), and metal accumulation impacts on benthic organisms (Baker and Yousef 1995) are evaluated. As important as structural aspects are to the success of engineered wetpond and wetland ecosystems, functional processes also are important, given that both structural and functional processes interact, and knowledge of both processes yields a more comprehensive understanding of ecosystems (Gessner and Chauvet 2002). Understanding the structure and function of naturally occurring wetlands and wetponds will allow engineers and biologists to incorporate that knowledge into the design, operation and expectations of engineered systems (Wetzel 1993).

The decomposition of leaf litter is an important functional process in the determination of the ecological condition of aquatic ecosystems (Suberkropp and Klug 1976, Cummins and Klug 1979, Suberkropp and Klug 1979, Webster and Benfield 1986, Maloney and Lamberti 1995, Gessner and Chauvet 2002). Leaves that fall in water are colonized by aquatic fungi and bacteria, which initiate the decomposition process and prepares leaf material for fragmentation by invertebrate shredders (Cummins and Klug 1979, Webster and Benfield 1986, Gessner and Chauvet 1994). The rate at which leaf detritus loses weight is due to characteristics of the leaf itself, as well as the structural and functional characteristics of the aquatic environment. High rates of detritus processing are associated with warm temperatures and availability of nitrogen and phosphorus (Meyer and Johnson 1983, Melillo et al. 1984, Jenkins and Suberkropp 1995, Suberkropp and Chauvet 1995) and aquatic macroinvertebrates (Benfield et al. 2001). These structural aspects, along with functional processes such as leaf litter breakdown can be used to assess the integrity of an aquatic system (Gessner and Chauvet 2002).

The City of Austin, Texas became interested in using wetponds for stormwater treatment in the 1980's as part of the Nationwide Urban Runoff Program (NURP). An existing detention pond, Woodhollow Pond, was modified to create a permanent pool, called a wetpond (Glick and Chang 1998). Austin installed its first wetpond, St. Elmo wetpond, in 1993 and many wetponds since, and has modified the City of Austin Land Development Code (COA 1997) to provide a wetpond option to land developers wishing to build stormwater detention basins that are also aesthetically pleasing. The City of Austin publishes its constructed wetpond design criteria in the Environmental Criteria Manual (COA 1997). The criteria address general and specific wetpond requirements, including the previously mentioned hydraulic residence time, vegetative shelf and wetpond design. Based on Glick and Chang (1998), the Environmental Criteria Manual specifies a hydraulic residence time of 14 days minimum during the wettest month of the year. By identifying the wettest month of the year through historical data, and measuring runoff from the development site, the wetpond design can be calculated to meet the required hydraulic residence time.

Vegetation specifications include a list of wetland species planted at each site, including floating, submergent and emergent species. A vegetative shelf is required in the sediment forebay and in the main pool, with the shelf covering 20% of the total wetpond surface area. A shelf is required to be at least 3.05 m wide with a 5 to 15% slope, with maximum water depth over the shelf not to exceed 45.72 cm (18 in) (Glick and Chang 1998).

The Environmental Criteria Manual identifies design requirements for the wetpond permanent pool, extended detention pool and optional flood detention pool. Wetponds are required to not be any deeper than 2.44 m (6 ft), to avoid possible odor problems associated with anaerobic conditions (Glick and Chang 1998).

This study examined the functional process of leaf detritus processing among eight different wetponds of varying age, watershed characteristics, water chemistry, and design specifications. Comparing leaf detritus processing among wetponds can aid in understanding how abiotic factors affect each wet pond as an ecosystem. The specific objectives were to determine: 1) factors or combinations of factors that affect detritus processing rates among wetponds, 2) characterize stages of succession among wetpond vegetation communities, 3) characterize water chemistry characteristics among wetponds 4) identify aquatic macroinvertebrate community differences among wetponds, and 5) compare the detritus processing rates of two plant species, *Platanus occidentalis* and *Carya illinoensis*.

SITE DESCRIPTIONS

The City of Austin is located in Travis and Williamson counties, Texas (Fig. 1). Annual rainfall is bimodal, with peaks in late spring and September, though there is extreme variability. Spring rains are characteristically of short duration and high intensity while winter rains occur as light showers (Werchan et al. 1974). Eight wetponds in Austin were chosen for this study (Fig. 2). Two exist in residential areas, three are part of commercial developments and three others occur in areas of mixed use.

Davis Spring wetpond (N 30° 29.763', W 097° 46.174') is a residential wetpond located in northwest Williamson County (Fig. 3). Construction and planting was completed in 1999. Permanent pool volume is 3466 m^3 (2.81acre-feet), and covers an area of 0.35 ha at permanent pool volume. The wetpond receives runoff from a 19.76 ha residential development, 5.74 ha is impervious cover.

Canterbury Trails wetpond (N 30° 09.317', W 097° 49.603') is a residential wetpond located in south Travis County (Fig. 4), and was constructed and planted in 1999. Permanent pool volume is 3453 m^3 (2.80 acre-feet), and covers an area of 0.30 ha at permanent pool volume. The wetpond receives runoff from a 33.05 ha residential development, 11.67 ha is impervious cover.

Motorola wetpond (N 30° 27.418', W 097° 45.005') is a commercial wetpond located on the Motorola campus in north Williamson County, and was constructed and planted in 1998 (Fig. 5). Permanent pool volume is 17550 m^3 (14.40 acre-feet), and covers an area of 0.35 ha at permanent pool volume. The wetpond receives runoff from a 13.06 ha commercial development, 9.31 ha is impervious cover.

National Instruments wetpond (N 30° 24.445', W 097° 43.425') is an in-stream commercial wetpond located on the National Instruments campus in north central Travis County (Fig. 6) and was constructed and planted in 2000. Pond permanent pool volume is 5032 m^3 (4.08 acre-feet), and covers an area of 0.27 ha at permanent pool volume. The wetpond receives runoff from a 213.27 ha commercial development, 25.59 ha is impervious cover.

St. Elmo wetpond (N 30° 12.437', W 097° 45.134') is a commercial wetpond located in southeast Travis County (Fig. 7). This wetpond was an experimental pond, from which design criteria were later modified to create the City of Austin's current wetpond design criteria. St Elmo wetpond was the first wetpond built in Austin, constructed in 1993. The pond is 2.44 m (8 ft) deep, while current design criteria requires a maximum 1.83 m (6 ft) depth. Permanent pool volume is 5550 m^3 (4.50 acre-feet), and covers an area of 0.77 ha at permanent pool volume. St. Elmo wetpond receives runoff from a 10.97 ha commercial development, 7.13 ha is impervious cover.

Upper Shoal Creek wetpond (N 30° 22.835', W 097° 44.079') is a mixed-use wetpond located in central Travis County (Fig. 8) and was constructed and planted in 1998. Permanent pool volume is 17392 m^3 (14.10 acre-feet), and covers an area of 0.35 ha at permanent pool volume. The wetpond receives runoff from a 406.33 ha mixed use development, 325.06 ha is impervious cover.

Central Park wetpond (N 30° 18.230', W 097° 44.301') is a mixed-use wetpond in central Travis County (Fig. 9) and was constructed and planted in 1995. Permanent pool volume is 3453.75 m^3 (2.80 acre-feet), and covers an area of 0.30 ha at permanent pool volume. The wetpond receives runoff from a 33.05 ha mixed-use development, 11.67 ha is impervious cover.

Alpine wetpond (N 30° 13.660', W 097° 45.751') is a mixed-use wetpond located in south Travis County (Fig. 10), and was constructed and planted in 1998. Permanent pool volume is 1386 m^3 (1.12 acre-feet), and covers an area of 0.16 ha at permanent pool volume. The wetpond receives runoff from a 23.30 ha mixed-use development, 8.16 ha is impervious cover.

METHODS

Leaf litter was collected from pecan (*Carya illinoensis*) and sycamore (*Platanus occidentalis*) trees on the Southwest Texas State University campus in San Marcos, Hays County, Texas. Leaves were collected from October through November after abscission and before the onset of decomposition by using a tarp to catch falling leaves. Leaves were dried in a Fisher Scientific Isotemp Oven (Model 655 F) at 50° C for at least 24 h. Only leaves that did not show evidence of herbivory were selected for the study. Leaves were weighed to as close to 4 g as possible in a Mettler AT100 balance and enclosed in sealed plastic bags until placement into litter bags. Plastic mesh (1 x 1.5 mm) screen was used to construct 15 x 10 cm litter bags. Three to four holes (0.07 cm) were cut into both sides of each bag to allow access to larger aquatic macroinvertebrates. Pre-weighed (4g) leaves were transferred from plastic bags to litter bags, and secured shut with staples. Two bags, each containing one type of leaf litter were tied together and then secured to a house brick with 25-lb strength nylon monofilament line. Four to five leaf litter bag pairs were tied to each brick. Bricks with leaf packs were placed on the vegetative shelf/littoral zone of each wetpond. Locations too close to an influent or effluent point were avoided to minimize loss of bags during rain events. A set of handling-loss bags was used to correct for leaf weight lost due to handling and transport. The handling-loss set of bags were submersed in the wetpond at each location for several seconds, removed, enclosed in a sealed bag and placed on ice for transport to the lab. Results from the handling-loss set were used as the starting point from which to measure weight loss during the course of the study. In the laboratory after collection, leaves were removed from each bag, dried for a minimum of 24 h at 50° and dry mass was weighed with a Mettler AT100 balance. Leaf litter bags were collected from each of the 8 wetponds after 7, 14 and 28 d, and monthly thereafter for five months. Dry weight of leaf litter may be affected by the accumulation of sediment and lead to underestimates of breakdown rates (Gessner and Chauvet 1994, Benfield 1996). For this reason, ash-free dry mass values were also determined (Benfield 1996). Leaves from each sample site were ground and a 1 g sample was ignited at 550 °C for 24 h.

Based on preliminary observations, it was determined that pecan leaves would decay more rapidly than sycamore. For this reason, pecan leaves remained in the wetponds for 113 days, while sycamore leaves remained for 205 days. Consequently, discussion involving the order of decomposition rates mainly emphasizes sycamore decomposition rates, and the order within which sycamore decomposed within the wetponds.

Diel studies at Davis Spring, Motorola and Central Park wetponds measured dissolved oxygen and temperature with an Orion Model 842 Meter, and pH with a Fisher Scientific Accumet AP85 Meter every two hours over a 24-hour period. Measurements were taken on the vegetative shelf, at the soil/water interface, to partially characterize the physical and chemical environment within which the leaf litter bags were placed.

Aquatic macroinvertebrates collected from the litter bags were preserved in 70% ethanol, counted and identified to family using keys by Merritt and Cummins (1996). Taxonomic richness was determined only for macroinvertebrates collected from litter bags of each wetpond.

Plant surveys were conducted to provide a qualitative record of floating, emergent and submergent species present at each wetpond. Surveys were conducted during the summer field collections. A plant survey was not conducted for Central Park and National Instruments wetponds because these wetponds are highly maintained, and thus successional changes in species composition or successful invasions by exotics were unlikely. A then and now comparison of vegetation provided information on persistence of initial plant communities in different wetponds and successional changes. Vegetative similarity between plant composition at the time of wetpond establishment and this study was calculated using Sorensen's Index of Similarity where $S = [2C/(s_1 + s_2)] \times 100$, where C = number of species common to both communities and s_1 and s_2 = number of species in communities 1 and 2 (Smith 1986).

Principal component analysis (PCA, Zar 1999) was used to identify if wetpond variables affected detritus processing rates. Wetponds with the highest detritus processing rates were compared for morphometric similarities that could highly affect detritus processing.

Leaf detritus processing rates were analyzed using exponential regression analysis where $m_t = m_o e^{-kt}$, such that m_t is the mass remaining at time t, m_o is the initial mass, and k the breakdown coefficient (Wider and Lang 1982, Gessner and Chauvet 1994). The slopes of sycamore and pecan detritus processing rates were statistically compared with analysis of variance (ANOVA). Decomposition rates were modeled using exponential and linear regression. Differences between results were analyzed using analysis of covariance (ANCOVA) because data from both analyses produced similar results. Estimates of breakdown rates were similar so results were presented in linear regression.

Significant differences in detritus processing rates between pecan and sycamore in Group 1 and Group 2 wetponds were analyzed using multiple regression analysis.

RESULTS

Abiotic factors

Water temperatures in all wetponds monitored for 24 h followed typical diel patterns; water temperatures were higher during afternoon and evening hours and were lower during the night and early morning (Fig. 11). Dissolved oxygen levels were nearly anoxic (i.e., <0.5 mg/L) with slight increases observed during the afternoon. For pH, Upper Shoal Creek and Motorola wetponds followed typical diel patterns (e.g., higher pH with increased temperatures) whereas pH in Davis Spring wetpond remained constant over a 24-hour period.

Invertebrate assemblage

Macroinvertebrate taxa richness associated with leaf litter bags ranged from a total of 12 to 6 taxa (Table 1). Macroinvertebrate abundance was generally low. The majority of macroinvertebrates collected at all wetponds consisted of Diptera, Chironomidae and Oligochaeta species. Trichoptera species were collected only from two wetponds, and four wetponds showed a dominance of Oligochaeta. Two wetponds showed an abundance of Diptera, and one wetpond showed a dominance of Odonata. There does not appear to be any trends between detritus processing rates and aquatic macroinvertebrates collected for each of the wetpond sites.

Plant Composition

Plant composition comparisons between initial planting and this study showed that persistence of initial plant species was greatest at Davis Spring (82%), followed by

Canterbury Trails (64%), Motorola (60%), Alpine (36%) and finally Upper Shoal Creek (31%) wetponds (Table 2). Sorensen's Index of Similarity values were greatest at Canterbury Trails (61%), then Motorola and Davis Spring (60%), Alpine (48%) and finally Upper Shoal Creek (43%). Similarity could not be determined for St. Elmo wetpond because the initial planting list has not been located. At each wetpond site, several species planted at construction were not present during the survey, specifically, fanwort (*Cabomba caroliniana*) and some *Scirpus* species. All wetponds except Upper Shoal Creek had varying amounts of either cattail or hydrilla, or both. The oldest wetpond, St. Elmo, was heavily dominated by cattails.

Using principal component analysis (PCA), morphometrics explained 98% of the observed variation among wetponds (Table 3). Figure 12 shows a scatterplot of the PCA results, and a table representing the respective loadings of each morphometric variable. The analysis divided wetponds into two distinct groups on either side of PCA axis 1, which described an age and impervious cover gradient. Motorola, Upper Shoal Creek and Central Park wetponds had high scores along the axis for high impervious cover (50), and older age wetponds (48). These three wetponds later are identified as Group 1 in the multiple regression analysis. National Instruments, Alpine, Canterbury Trails and Davis Spring wetponds had low scores, and are identified as Group 2 in the multiple regression analysis. The PCA axis 2 described a permanent pool volume and watershed area gradient, and no significant relationships were found.

Initially, wetpond use type (commercial, residential, mixed-use) was included in the PCA, but relationships were insignificant and the variable was removed. Leaf litter bags at St. Elmo were found periodically out of water and dry so processing received by leaves at this wetpond was not comparable to processing of leaves at all other wetponds in the study. For this reason, St. Elmo wetpond was considered an outlier in the PCA, and was excluded from the multiple regression analysis.

Detritus processing rates showed distinct variations among the eight wetpond sites and between the two leaf types (Fig 12). Exponential regression results ranged from k = 0.00052 ($P < 0.0001$) to 0.00021 ($P = 0.0136$) for sycamore and from k = 0.00172 ($P < 0.0001$) to k = 0.00048 ($P < 0.0001$) for pecan (Table 4).

A multiple regression analysis found significant differences in pecan and sycamore detritus processing rates between Group 1 and Group 2 wetponds ($t = 3.77$, $P = 0.0002$, $df = 1$, $t = 2.23$, $P = 0.0267$, $df = 1$, Fig. 14), respectively. Detritus processing occurred more rapidly in Group 1 than in Group 2 wetponds, confirming that values of the morphometric variables shared within each of these groups resulted in similar detritus processing rates.

DISCUSSION

Overall detritus processing rates (k) in Austin wetponds were lower than literature k values for the same or similar species, and were generally lower than detritus processing rates for most leaf types (Gessner and Chauvet 1994, Petersen and Cummins 1974). Leaf processing rates below $k = 0.005$ are "slow" (Petersen and Cummins 1974). Slow rates occurred for both sycamore and pecan at almost all wetpond sites. Slow rates of detritus processing are generally attributed to leaf chemical constituents, water chemistry characteristics, and aquatic ecosystem characteristics (Webster and Benfield 1986). For St. Elmo wetpond, slow rates of detritus processing are also attributed to minimal exposure of leaf packs to water (see Brinson et al. 1981). Factors associated with urban landscapes and development such as sedimentation or pollution may add to these factors and lead to lower detritus processing rates.

The consistently higher decay rate of pecan relative to sycamore across all wetponds was most likely due to differences in leaf chemical constituents. Leaf nitrogen content (Webster and Benfield 1986, Taylor et al. 1989), and lignin and tannins (Gessner and Chauvet 1994) are used sometimes to predict decomposition. Greater initial leaf nitrogen content results in faster breakdown rates (Taylor et al. 1989), while higher concentrations of lignin and tannins retard decomposition (Gessner and Chauvet 1994). Sycamore leaves have a heavy leaf cuticle (McArthur and Marzolf 1987), and leaf lignin content ranges from 18.2% (Sharpe et al. 1980) to 30.9%

(Gessner and Chauvet 1994). These factors can affect the amount of material that leaches from leaves. Pignut hickory (*Carya glabra*), a congeneric of pecan, only has a leaf lignin content 10% (Suberkropp et al. 1976). Microbes show preferences for certain leaf species due to leaf chemical constituents (Gessner and Chauvet 1994), and sycamore is colonized more slowly than other species (Findlay and Arsuffi 1989, Gessner and Chauvet 1994). However, microbial growth on decaying leaves is positively affected by nutrient content of water (Jenkins and Suberkropp 1995, Bermingham et al. 1996), and so varying results for a single leaf species may occur in aquatic environments of different chemical characteristics.

This study was conducted in wetponds dominated by lentic conditions whereas most published studies of detritus processing are conducted in streams, dominated by lotic flow characteristics. Differences between streams and wetlands include dissolved oxygen availability, temperature, pH, nutrient availability and other factors (Polunin 1984, Tillman et al. 2003). Wetlands exhibit fluctuations in water levels, which differentially affect decomposition rates, nutrient recycling rates and system productivity (Day 1982) when compared to the stream environment. Leaf burial in wetlands due to sedimentation results in anaerobic conditions, further retarding decomposition (Brinson et al. 1981).

Sycamore had the lowest detritus processing rates at St. Elmo and highest detritus processing rates at Upper Shoal Creek. St. Elmo leaf litter bags were found dry on a majority of the field visits. At Upper Shoal Creek wetpond, some litter bags were found occasionally floating at the water surface or dry on the vegetative shelf after water receded. The wet-dry exposure experienced by leaf litter bags at Upper Shoal Creek wetpond may lead to more rapid detritus processing rates (Brinson et al. 1981), carbon and nitrogen release (Polunin 1984), and increased microbial activity (Sorensen 1974). St. Elmo wetpond consistently had low water levels throughout the study, and litter bags spent more time out of water than submerged; when found submerged, litter bags were completely buried in sediment. Although alternate wetting and drying conditions may lead to increased detritus processing rates, alternations between aerobic and anaerobic conditions, such as what occurred at St. Elmo wetpond, may result in lower rates of loss (Brinson et al. 1981).

The PCA indicates that wetponds with a higher amount of impervious surface area in the watershed, and older wetponds have higher detritus processing rates. I did not expect impervious cover to be positively related to higher detritus processing rates. Impervious cover results in higher sediment loading (Waters 1995, Sponseller and Benfield 2001, Roy et al. 2003), as well as in influxes of debris, litter, oils, heavy metals, nutrients and organic matter (Davies and Bavor 2000), and higher flow velocities (Webster and Benfield 1986), factors generally associated with lower detritus processing rates. Among other impacts, these pollutants lead to decreased water quality and alter macroinvertebrate assemblages (Sponseller and Benfield 2001, Roy et al. 2003). I assumed that these factors would be of importance in wetponds with high impervious cover, but did not measure them qualitatively or quantitatively. Alternatively, of the three wetponds associated with high impervious cover and faster detritus processing rates, Central Park and Motorola wetponds are landscaped areas, and fertilizers are used regularly. Thus, fertilizers may have enhanced decay (Meyer and Johnson 1983, Suberkropp and Chauvet 1995, Benfield et al. 2001). Upper Shoal Creek wetpond is not consistently maintained. Results of detritus processing at this wetpond may be due to a combination of higher impervious cover and wetpond age together with the factors discussed above.

Wetpond age was considered one of the important variables that would affect detritus processing rates during this study, and was identified as a major factor in the PCA. Although St. Elmo was the oldest wetpond in the study, it had the lowest detritus processing rates. As mentioned, this was due to a consistently low water level throughout the study, and that leaf litter bags were rarely in water. Central Park wetpond, the next oldest wetpond, had higher detritus processing rates and the order of detritus processing rates generally corresponded to wetpond age, after impervious cover. A follow-up study may provide further support that greater decomposition rates occur in older wetponds, especially if that study succeeds in keeping leaf litter bags at St. Elmo submerged.

Surface area to volume ratio, permanent pool volume, wetpond use type and watershed area variables were all initially considered to be a factor that affected detritus processing. These four factors were eliminated for various reasons. Surface area to volume ratio of wetponds was initially considered as a factor affecting detritus processing for two reasons. First, a high SA:V suggests more water may be in contact with the sediment surface, and therefore increased water treatment and particulate settling. Second, a high SA:V also suggests more water is in contact with the air and greater oxygenation potential. Accurate assessment of this feature involves implementing the use of tracer-dye techniques or the study of water flow patterns and stormwater residence time (see Walker 1998).

Permanent pool volume was eliminated from the study because leaf litter bags were placed only on the vegetative shelf, rather than at various water depths within the water body. Factors influential to decomposition such as dissolved oxygen, light, temperature, water velocity, and other factors, vary at different depths within the aquatic environment (Wetzel 2001). A study including an assessment of detritus processing at stratified depths within the entire water body, rather than just the vegetative shelf, would provide more information about the structure and function of a wetpond.

No significant differences in detritus processing rates were found among different wetpond use types in the PCA. At the onset of the study, I believed that wetpond use type could be an important variable in detritus processing rates, because the watershed areas of different use wetponds would produce runoff consisting of different water pollutants that significantly affect detritus processing. A study focusing on water chemistry characteristics, watershed impervious cover, wetpond vegetation, wetpond use type and runoff pollutants would more appropriately assess the effectiveness of vegetative buffers or other ground cover against sediment and pollutant loading.

Watershed drainage area was not a significant variable for this study. Design criteria require the constructed wetpond volume to retain runoff from the entire watershed of the developed site no matter the size. Variations in wetpond influent structural design do affect water flow patterns, and consequently the effectiveness of wetponds to treat runoff water (Walker 1998). A biological study of water chemistry, residence time and detritus processing rates within wetponds with different influent structures may further aid in assessing the best structural design for drainage influents (see Tchobanoglous 1993).

Dissolved oxygen, water temperature and pH affect leaf detritus processing rates (Brinson et al. 1981), and diel values could fluctuate strongly over a 24-hour period within these wetponds (Howard-Williams et al. 1989, Sellers et al. 1995). Dissolved oxygen readings were highest and also most variable at Upper Shoal Creek wetpond, where sycamore detritus processing was greatest. Temperature did not vary considerably among wetponds, although temperatures at Upper Shoal Creek wetpond were consistently warmer than either Davis Spring or Motorola wetponds. Detritus processing may have been more rapid at Upper Shoal Creek due to favorable conditions in these parameters. For all wetponds, temperature was highest during late hours of the day. Diel pH values increased significantly during late evening and late morning hours at Upper Shoal Creek and Motorola wetponds, possibly in response to increases in photosynthetic activity associated with daylight and warmer temperatures (Polunin 1984). Temperature readings at Davis Springs did not change significantly.

A majority of macroinvertebrates collected at all wetponds consisted of Diptera, Oligochaeta, Hirudinea, and Odonata. Chironomidae (Order Diptera) were the most numerous aquatic insects collected, second only to Hirudinea. Chironomids are widespread, abundant food generalists (Merritt and Cummins 1996). Coenagrionidae (Order Odonata) were also abundant. Because they are predators (Merritt and Cummins 1996), this insect would not have contributed to leaf mass loss. Class Oligochaeta were very numerous, especially at National Instruments wetpond, where *k* values were extremely low (sycamore) to average (pecan). Oligochaeta eat mostly by ingesting sediment (Brinkhurst and Gelder 2001), and therefore likely did not contribute to leaf mass loss. Order Hirudinea, leeches, generally persist in the benthos, do not feed on leaf material, and are tolerant of polluted environments (Davies and Govedich 2001). It is most likely that leaf mass loss in these wetponds primarily was due to microbial processing. Leaf breakdown is caused by abiotic factors such as leaching and physical abrasion by moving water, and by biotic factors such as shredder invertebrates, microbial decomposition and conditioning (Webster and Benfield 1986). These factors vary in their relative importance in different aquatic ecosystems. I found no shredder invertebrates, and a limited diversity of aquatic macroinvertebrates overall, eliminating this form of mechanical processing as an explanation for leaf breakdown. Constructed wetponds are enclosed, lentic environments. High velocity flows occur only during heavy rain events, minimizing mechanical processing due to water velocity and turbulence. Leaching occurred, however weight loss due to leaching predominately occurs in the first 24 h of leaf decomposition (Petersen and Cummins 1974). Fungi play an important role in the decomposition of leaf material, colonizing leaves, breaking down polysaccharides and potentially preparing leaf material for consumption by aquatic macroinvertebrates (Webster and Benfield 1986). Several papers (Suberkropp and Klug 1979, Gessner and Chauvet 1994, Jenkins and Suberkropp 1995) cite the influence and importance of microbial activities in detritus processing. A more thorough study of aquatic macroinvertebrate and microbial populations among wetponds would yield significant information about the structure and function of wetpond aquatic ecosystems, and would aid in further understanding the biological capabilities of wetponds for water treatment. It would be interesting to identify differences in microbial populations among streams, wetlands and constructed wetponds, for purposes of water treatment capabilities by microbial processes. Also, constructed wetponds would be an excellent environment in which to study microbial processing of leaf material, especially if thorough field sampling resulted in limited presence of aquatic macroinvertebrates and no shredder species.

Initial plant species persistence assessment was higher at younger wetponds, indicating that with time, wetpond sites were invaded by species not planted at construction. Results of the vegetation similarity index shows decreasing similarity in vegetation composition among wetponds as they age. Although my main focus was not on plant community development, this is an important area of research. Understanding the establishment and development of plant community structure and function, in combination with water quality sampling may suggest that plants other than those considered as invasive species play a positive role in how wetponds fulfill their anthropogenic purpose. Most studies of vegetation development in constructed wetlands and similar systems focus on plants for community development (Reinartz and Warne 1993, Keddy et al. 1994, Keddy 1999, Lopez and Fennessy 2002), more so than for their water treatment capabilities. Conversely, research involving use of plants to treat wastewater (such as in sewage effluent treatment ponds) study plant species only for their nutrient removal capabilities, and focus on a very small selection of plants, or on species that would be undesirable for use in wetponds, constructed wetlands, or similar features designed for mitigation, and stormwater runoff (see Brix 1993, Debusk et al. 1996, Frankenbach and Meyer 1999, Körner et al. 1998, Schaafsma et al. 2000). Neither area of research appears to study plant community competition and development along with the water treatment capabilities of a variety of wetland plant species. A thorough study combining these two areas of research would significantly further the use of constructed systems for storm water treatment, due to the fact that plant species known most for their water treatment capabilities are also those most invasive, tending to create monoculture communities. The ultimately successful constructed water treatment system would combine the treatment of water with a diverse wetland plant community.

SUMMARY

Wetland construction and mitigation is a hot area of study because natural wetlands are continuously being filled for construction purposes. The intent is to rebuild wetlands in protected areas where they can continue to provide their multiple benefits to humans and wildlife. However, researchers are discovering that they do not completely understand how to recreate wetlands, and many rebuilt wetlands are not providing the same benefits as the original, displaced wetland. We do not fully understand how they function.

The detrital pathway in any ecosystem is a map to understanding the structure and function of an ecosystem, whether it is terrestrial or aquatic. Understanding structural and functional development can help us create them successfully. E.P. Odum (1969) stated that, although ecological succession is a complex process, it always progresses toward homeostasis. The success of a constructed ecosystem is determined only with the passing of time, as they progress towards a structural and functional homeostasis. When do constructed ecosystems reach a homeostasis?

I attempted here to add more knowledge to this field of study. By comparing a relatively large number of wetponds, that varied structurally, I observed functional processes and compared these to structural features. I found that detritus processing, a functional process, may increase with impervious cover and age, both structural features. Also, detritus processing rates varied among wetponds of similar age, which indicates influences from other variables. There are many opportunities for study beyond the focus

of my research, and I stated them thoughout my thesis. Studies within this field would benefit those who design constructed wetlands and would produce significant contributions to the development of constructed wetlands. Information about the development of constructed wetlands, constructed wetponds and all constructed ecosystems and would benefit those who design constructed ecosystems for all purposes.

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Table 1. Relative abundance of invertebrate taxa collected from litterbags in Austin, TX wetponds from January through August 2002. TNTC= Too Numerous To Count.

	Davis Spring	Canterbury Trails	Motorola	National Instruments	Upper Shoal Creek	Central Park	Alpine	St. Elmo
	%	%	%	%	%	%	%	%
Phylum Mollusca								
Class Gastropoda								
Family Planorbidae		**8**		**2**	**8**			
Family Physidae					**22**	**2**	**2**	**2**
Phylum Annelida								
Class Oligochaeta	**40**	**5**	**3**	**TNTC**	**14**	**7**		**27**
Order Hirudinea	**21**	**66**	**8**	**27**	**15**	**10**	**9**	**53**
Phylum Arthropoda								
Subphylum Uniramia								
Order Diptera								
Family Chironomidae	**33**	**9**	**31**	**56**	**14**	**46**	**35**	**5**
Family Ceratopogonidae						**22**		
Order Odonata								
Family Libellulidae	**1**	**9**	**<1**	**1**	**1**		**2**	
Family Corduliidae			**<1**					
Family Macromiidae					**1**		**4**	
Family Coenagrionidae	**3**		**18**	**3**	**7**	**2**	**37**	
Order Ephemeroptera								
Family Caenidae	**1**		**3**	**1**				
Family Ephemeridae	**1**							
Family Tricorythidae			**2**					
Family Isonychiidae			**<1**					
Order Trichoptera		**3**					**2**	
Order Coleoptera							**2**	**3**
Family Haliplidae					**12**			**2**
Family Dystiscidae								**3**
Subphylum Chelicerata								
Class Arachnida			**<1**			**2**		
Subphylum Crustacea								
Division Eubranchiopoda								
Class Branchiopoda			**<1**			**6**	**7**	**5**
Order Ostracoda			**32**	**9**	**6**	**3**		
Order Cladocera				**1**				
Total number	**78**	**66**	**121**	**>95**	**73**	**69**	**46**	**59**

Table 2. Vegetation observed during summer sampling period at select Austin, TX wetpond sites.
C = species planted at construction S = species present during summer sampling period

		Davis Spring		Canterbury Trails		Motorola		Upper Shoal Creek		Alpine		St. Elmo	
Scientific Name	**Common Name**	C	S	C	S	C	S	C	S	C	S	C	S
Cabomba caroliniana	fanwort	x		x		x		x		x			
Carex sp			x										x
Ceratophyllum sp	coontail			x	x	x	x						
Ceratophyllum demersum	Coontail	x	x					x	x	x			x
Cyperus ochraceus	Flat sedge							x	x				
Cyperus odoratus	Rusty Flat sedge		x							x			x
Dichromena colorata	White Star sedge							x		x			
Echinodorus rostratus	burhead							x		x			
Eleocharis macrostachya	Creeping spikerush		x					x		x	x		x
Eleocharis montevidensis	Sand spikerush	x		x	x	x	x	x		x	x		x
Eleocharis quadrangulata	Four-square spikerush	x	x	x	x	x	x	x	x	x			x
Hydrilla verticillata	Hydrilla				x		x						
Hydrocotyle sp.	Water pennywort		x		x				x		x		x
Iris pseudacorus	Yellow Flag iris	x	x	x				x		x			x
Juncus sp	rush		x										x
Juncus effusus	Soft-stem bulrush					x		x		x	x		
Justicia americana	Water willow								x	x	x		
Lemna sp	duckweed				x								x
Lippia nodiflora	Frog fruit		x										x
Ludwigia repens	False loosestrife		x		x				x		x		
Marsilia macropoda	Water clover		x			x		x		x			x
Najas guadalupensis	Water-naiad					x		x	x	x			
Nymphaea odorata	Fragrant waterlily	x	x	x	x		x	x		x	x		
Pontederia cordata	pickerelweed	x	x	x	x			x		x	x		x
Potamogeton sp.	pondweed							x					
Potamogeton pectinatus	Sago pondweed	x	x	x			x			x			x
Rhynchospora corniculata	Beak rush							x		x	x		
Sagittaria sp	arrowhead							x	x				
Sagittaria latifolia	Arrowhead	x	x	x	x	x	x			x			x
Scirpus sp	bulrush							x	x				x
Scirpus californicus	California bulrush	x	x	x	x	x	x			x			
Scirpus pungens	Three-square bulrush		x					x		x			
Scirpus validus	Softstem bulrush	x	x	x		x	x			x			
Thalia dealbata	Powdery Alligatorflag									x	x		
Typha latifolia	Cattail		x		x		x				x		x
	Total	11	19	11	12	10	10	19	9	22	11	N/A	17
	Persistence (%)	0.81		0.63		0.60		0.32		0.36		N/A	
	Similarity Index	0.58		0.61		0.60		0.43		0.48		N/A	

Table 3. Principal component analysis morphometric variables. Table 3a identifies actual values of variables, Table 3b identifies variables as entered for analysis. Asterisk (*) identifies values that were estimated when actual values were not included in original site engineering plans.

3a.

Wetpond	Age	Surface Area at PPV (ha)	Use Type	Percent Impervious Surface Area	Permanent Pool Volume (m^3)	Watershed Area (ha)
Davis Spring	3	0.40	R	35	3,466	19.8
Motorola	4	0.35	C	71	17,551 *	13.1
National Instruments	2	0.27	C	12	5,033	213.2
Upper Shoal Creek	4	0.35	M	68	17,392	406.0
St. Elmo	9	0.77	C	65	5,551	11.0
Central Park	7	0.30	M	54	8,495	67.0
Canterbury Trails	3	0.30	R	35	3,454	33.0
Alpine	4	0.16	M	35	1,386 *	23.0

3b. Age (1-3 = 1, 4-6 = 2, 7-9 = 3), percent impervious surface area (0-35% = 1, 36-65% = 2, 66-71% = 3), use type (R = 100, C = 010, M = 001), watershed area (0-33 ha = 1, 34-70 ha = 2, 70-400 ha = 3).

Wetpond	Age	Surface Area at PPV	Use Type R	Use Type C	Use Type M	Percent Impervious Surface Area	Permanent Pool Volume	Watershed Area
Davis Spring	1	0.40	1	0	0	1	3,466	1
Motorola	2	0.35	0	1	0	3	17,551 *	1
National Instruments	1	0.27	0	1	0	1	5,033	3
Upper Shoal Creek	4	0.35	0	0	1	3	17,392	3
St. Elmo	3	0.77	0	1	0	3	5,551	1
Central Park	3	0.30	0	0	1	2	8,495	2
Canterbury Trails	1	0.30	1	0	0	1	3,454	1
Alpine	2	0.16	0	0	1	1	1,386 *	1

Litter Type	Wetpond	Exponential Regression *k*	*P*
Sycamore	St. Elmo	-0.00021	0.0136
	National Inst	-0.00033	<0.0001
	Canterbury	-0.00034	<0.0001
	Motorola	-0.00039	<0.0001
	Davis Spring	-0.00040	<0.0001
	Alpine	-0.00041	<0.0001
	C Park	-0.00044	<0.0001
	USC	-0.00052	<0.0001
Pecan	St. Elmo	-0.00045	0.0003
	Canterbury	-0.00071	<0.0001
	Davis Spring	-0.00086	<0.0001
	Alpine	-0.00099	<0.0001
	National Inst	-0.00116	<0.0001
	USC	-0.00129	<0.0001
	C Park	-0.00132	<0.0001
	Motorola	-0.00172	<0.0001

Table 4. Detritus processing rates (*k*) of pecan and sycamore leaves in constructed wetponds.

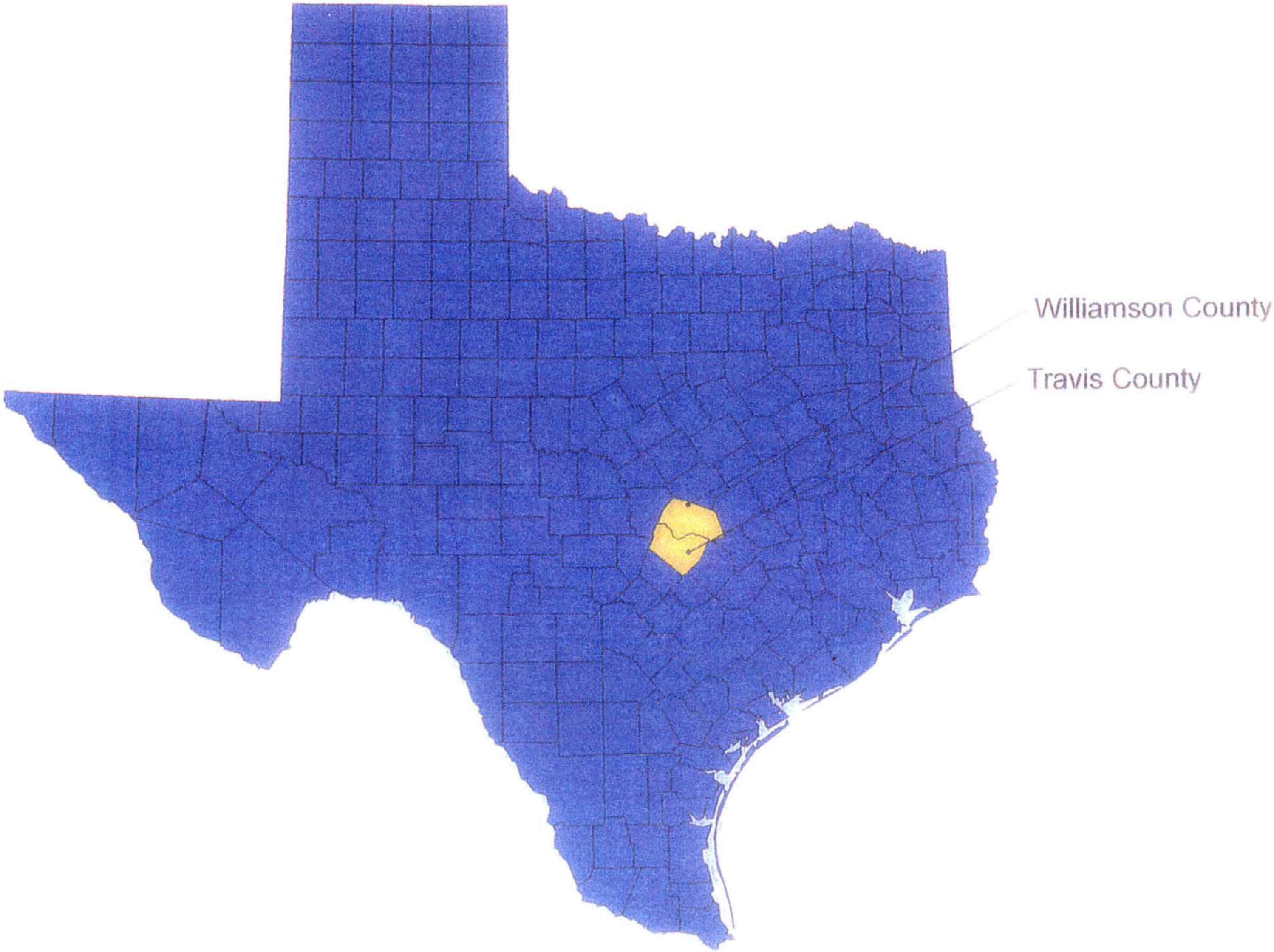


Fig. 1. Project site, Travis and Williamson counties, Texas

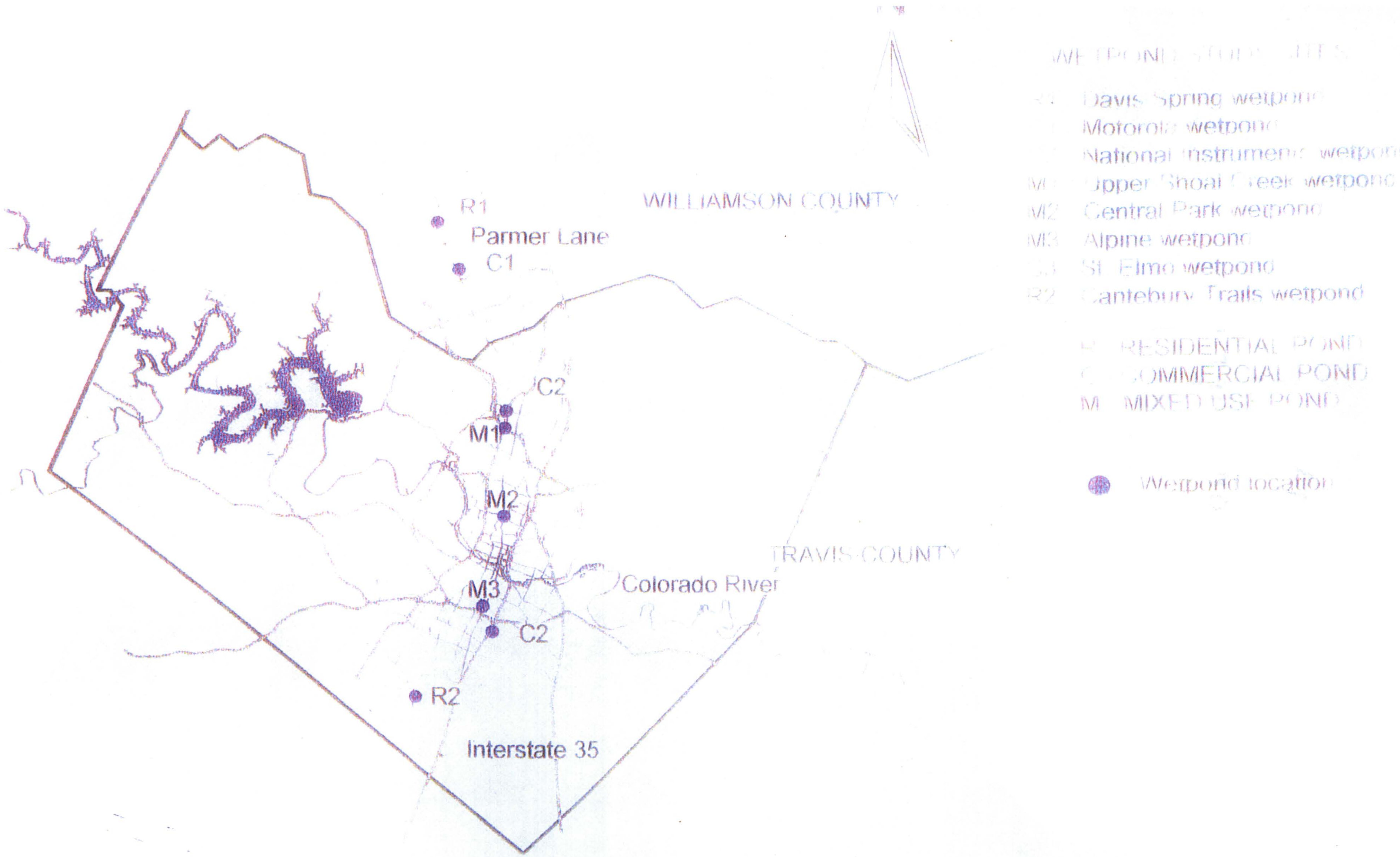


Fig. 2. **Map of wetpond study site locations in Austin, Texas**

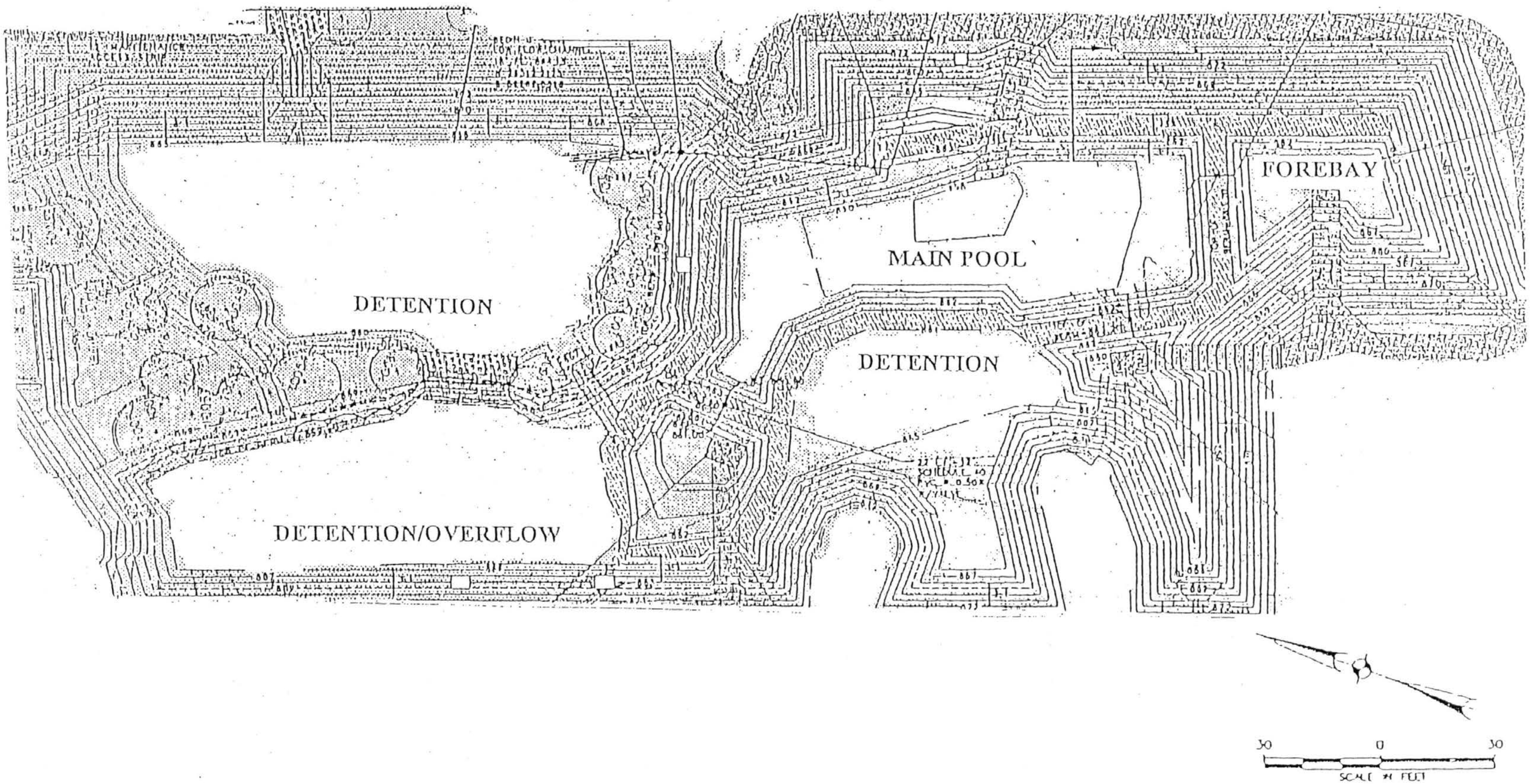
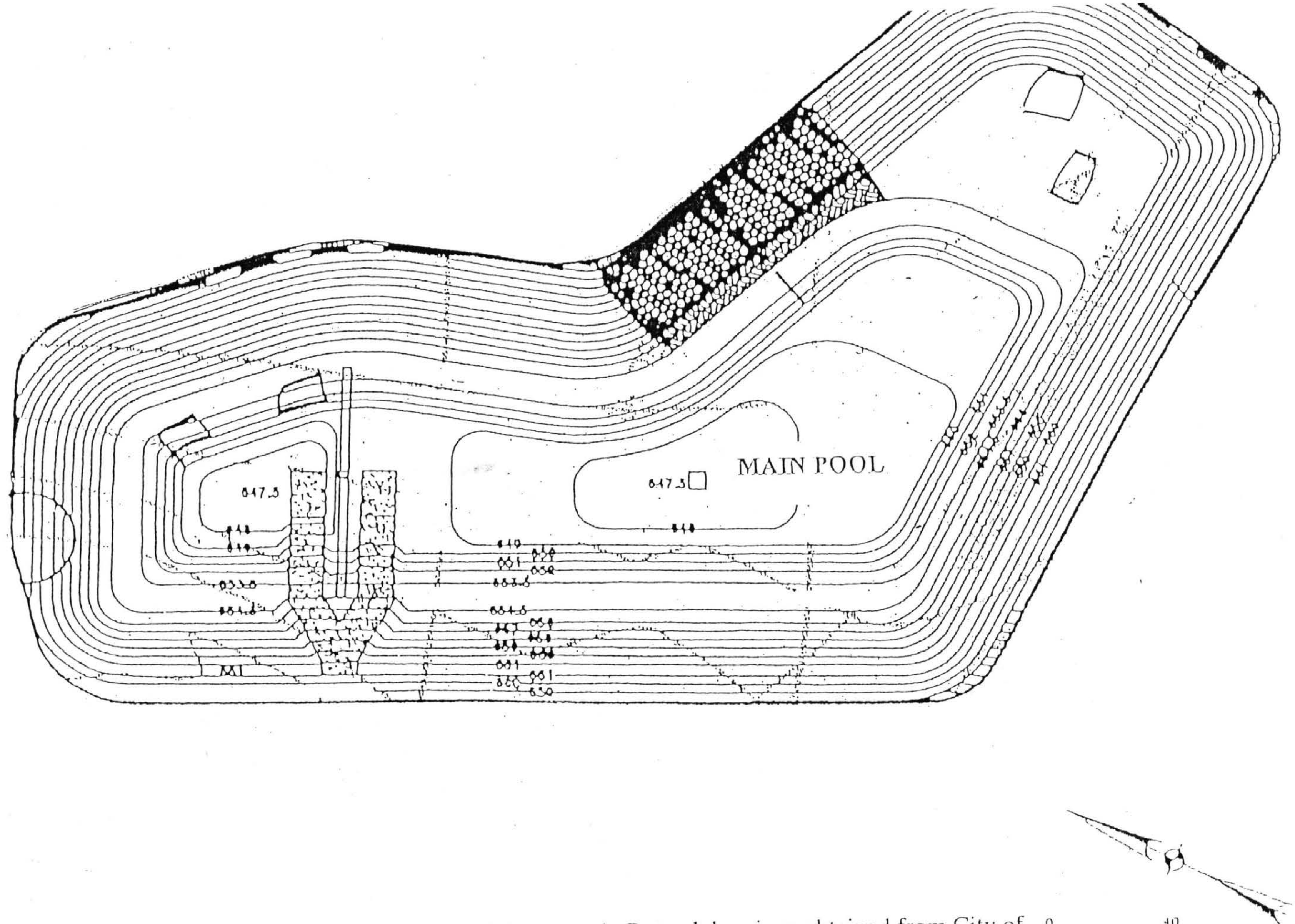


Fig. 3. Davis Spring residential wetpond. Record drawings obtained from City of Austin Development Assistance Center.



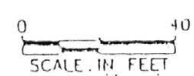


Fig. 4. Canterbury Trails residential wetpond. Record drawings obtained from City of Austin Development Assistance Center.

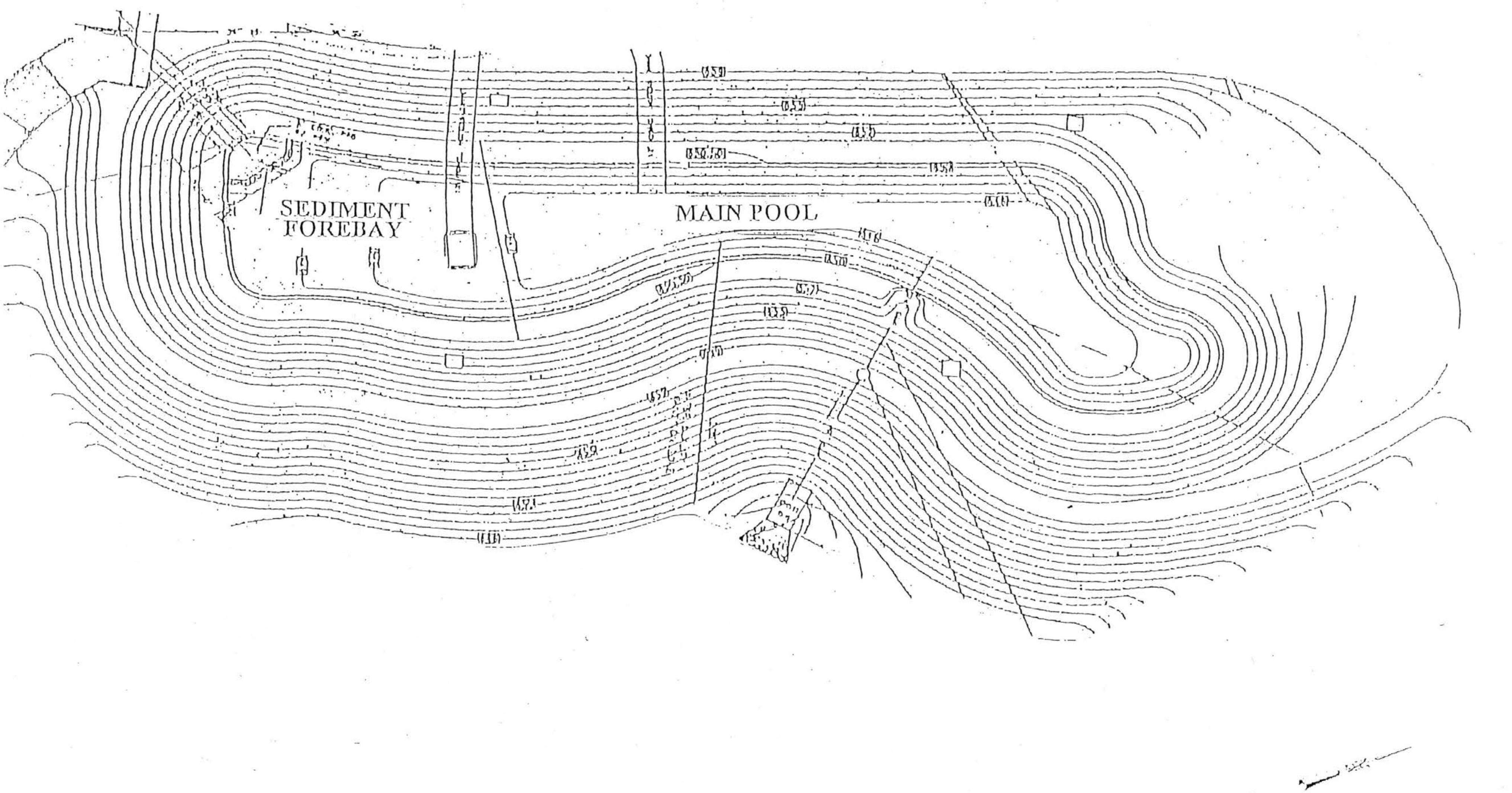


Fig. 5. Motorola commercial wetpond. Record drawings obtained from City of Austin Development Assistance Center.

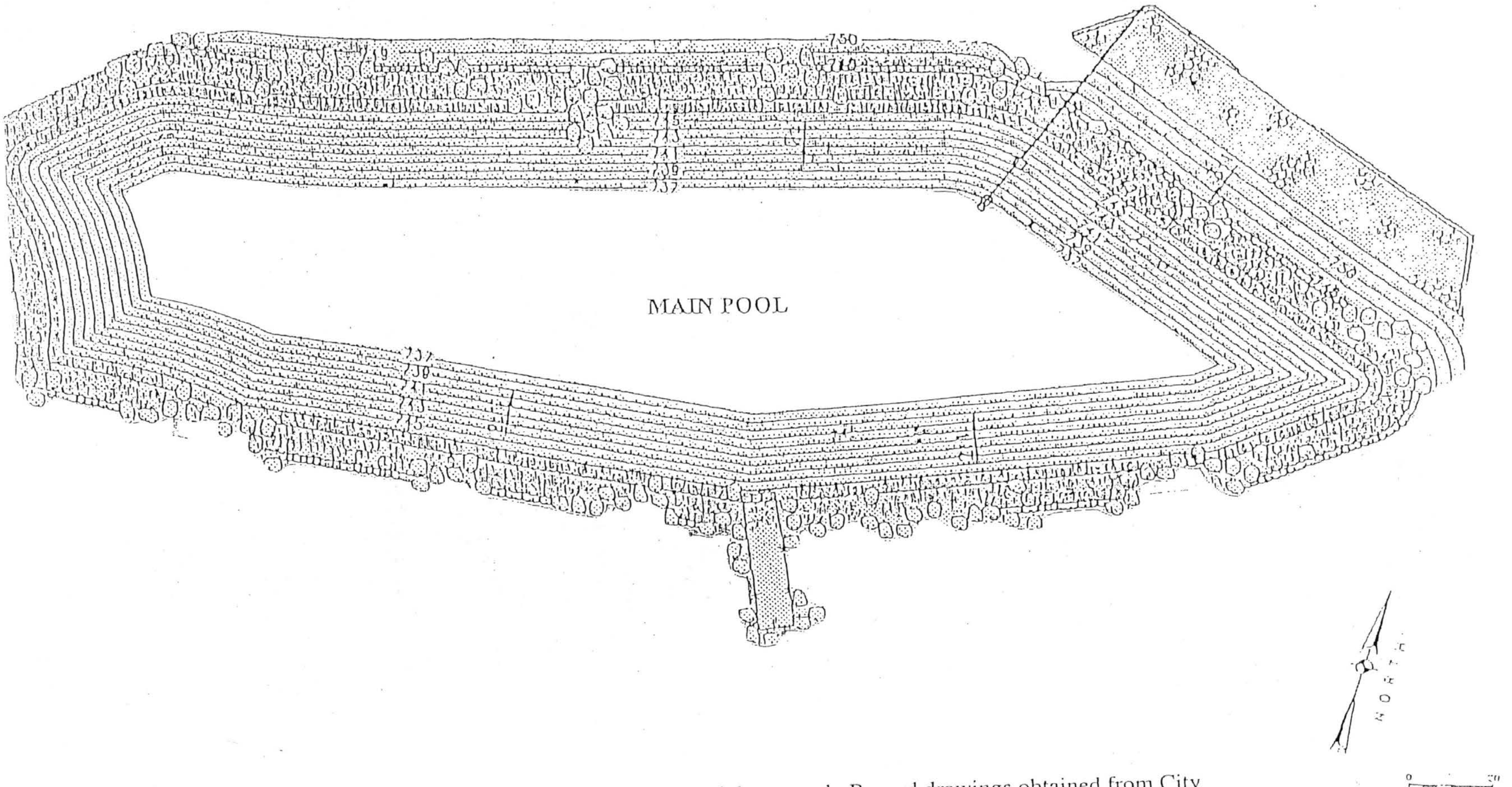


Fig. 6. National Instruments commercial wetpond. Record drawings obtained from City of Austin Development Assistance Center.

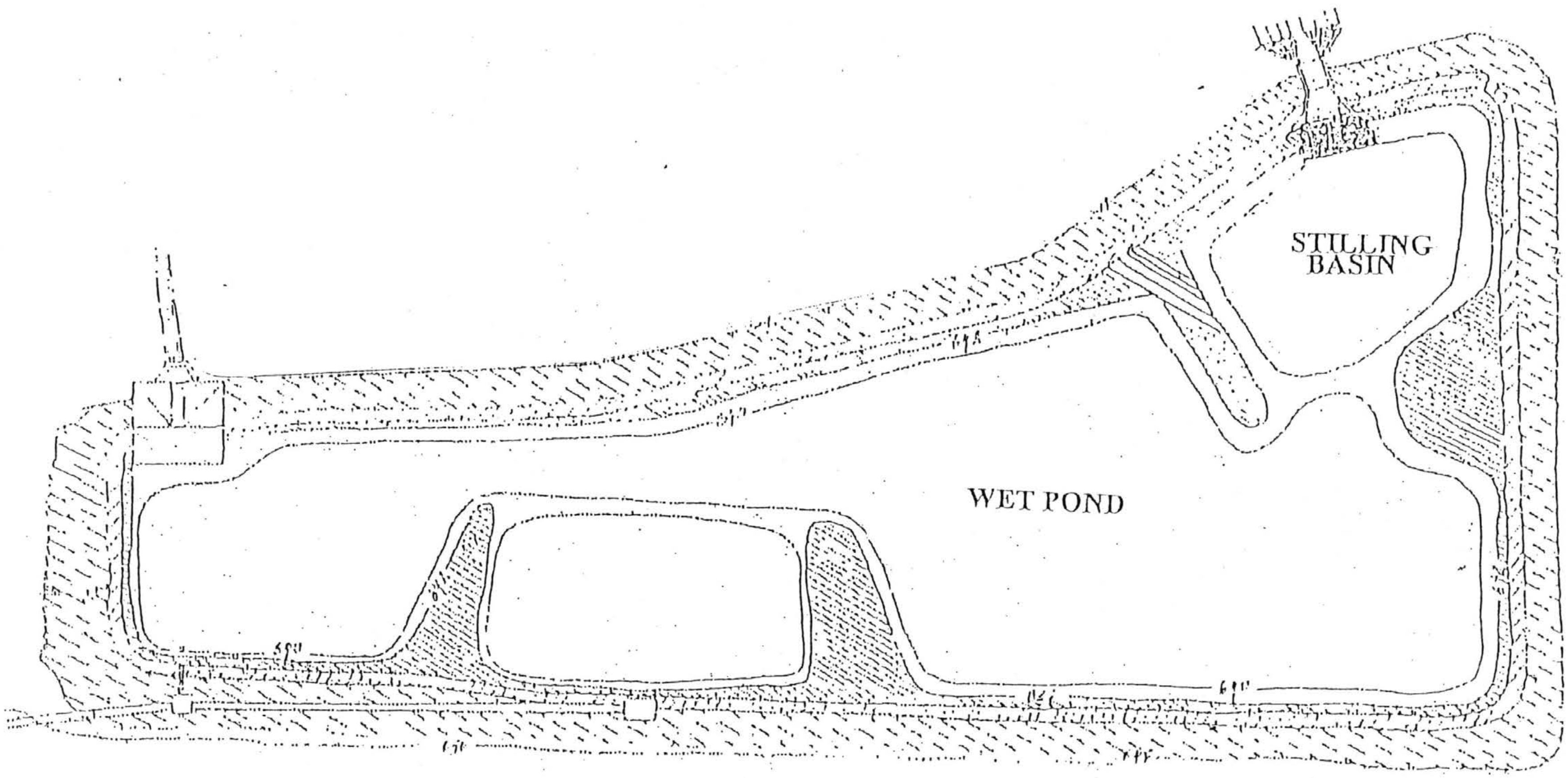


Fig. 7. St. Elmo commercial wetpond. Record drawings obtained from City of Austin Development Assistance Center.

SCALE: 1" = 30'

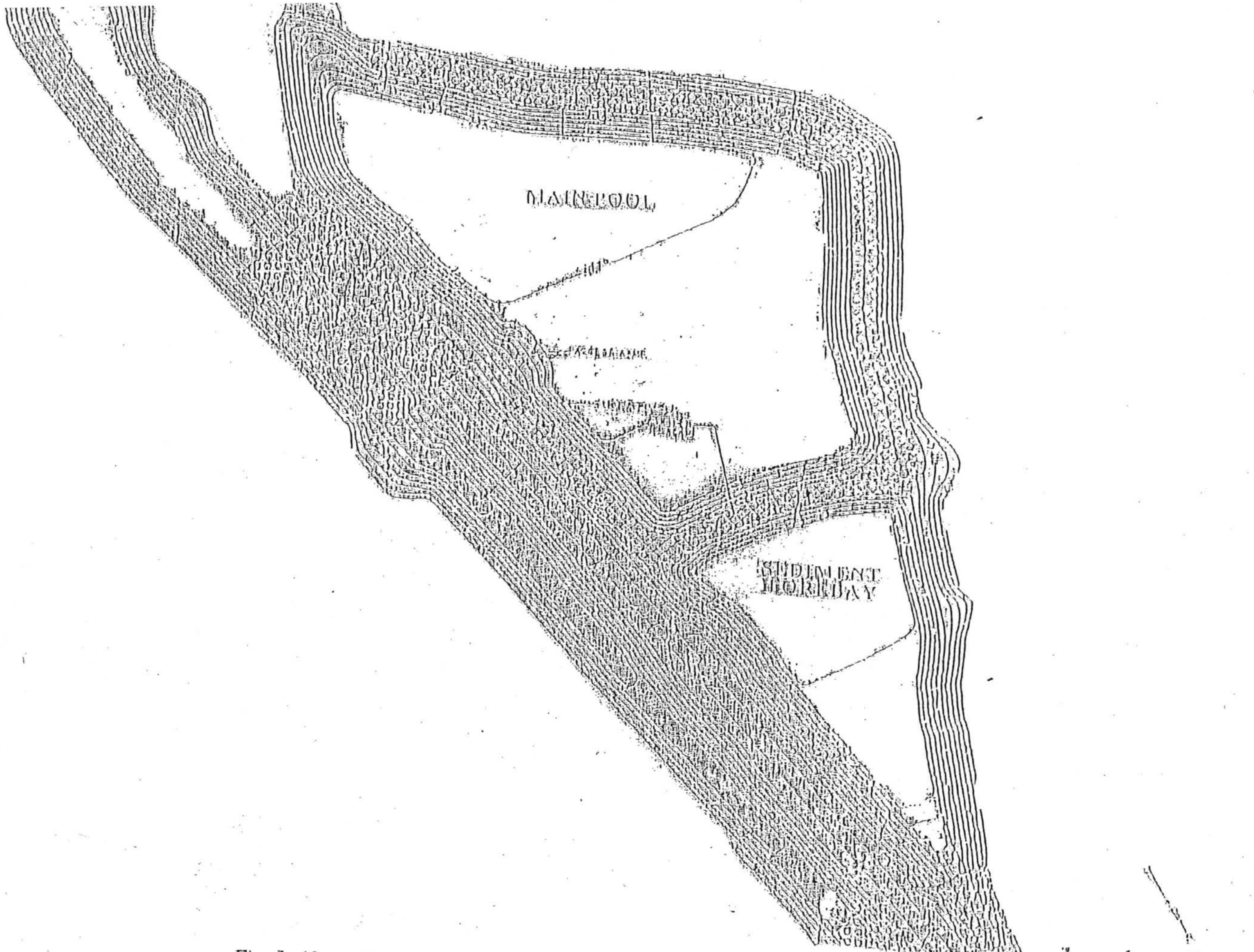


Fig. 8. Upper Shoal Creek mixed-use wetpond. Record drawings obtained from City of Austin Development Assistance Center.

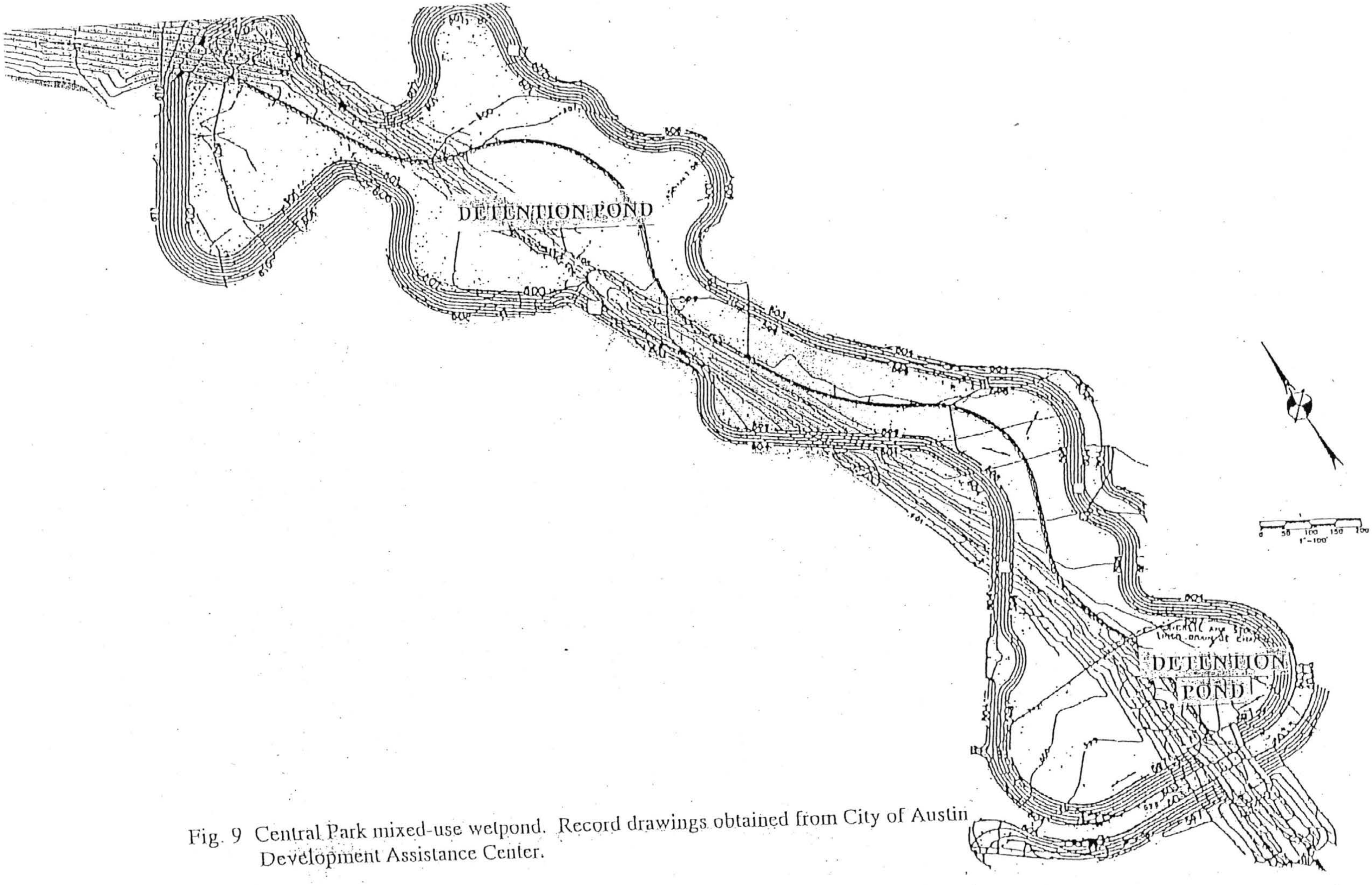
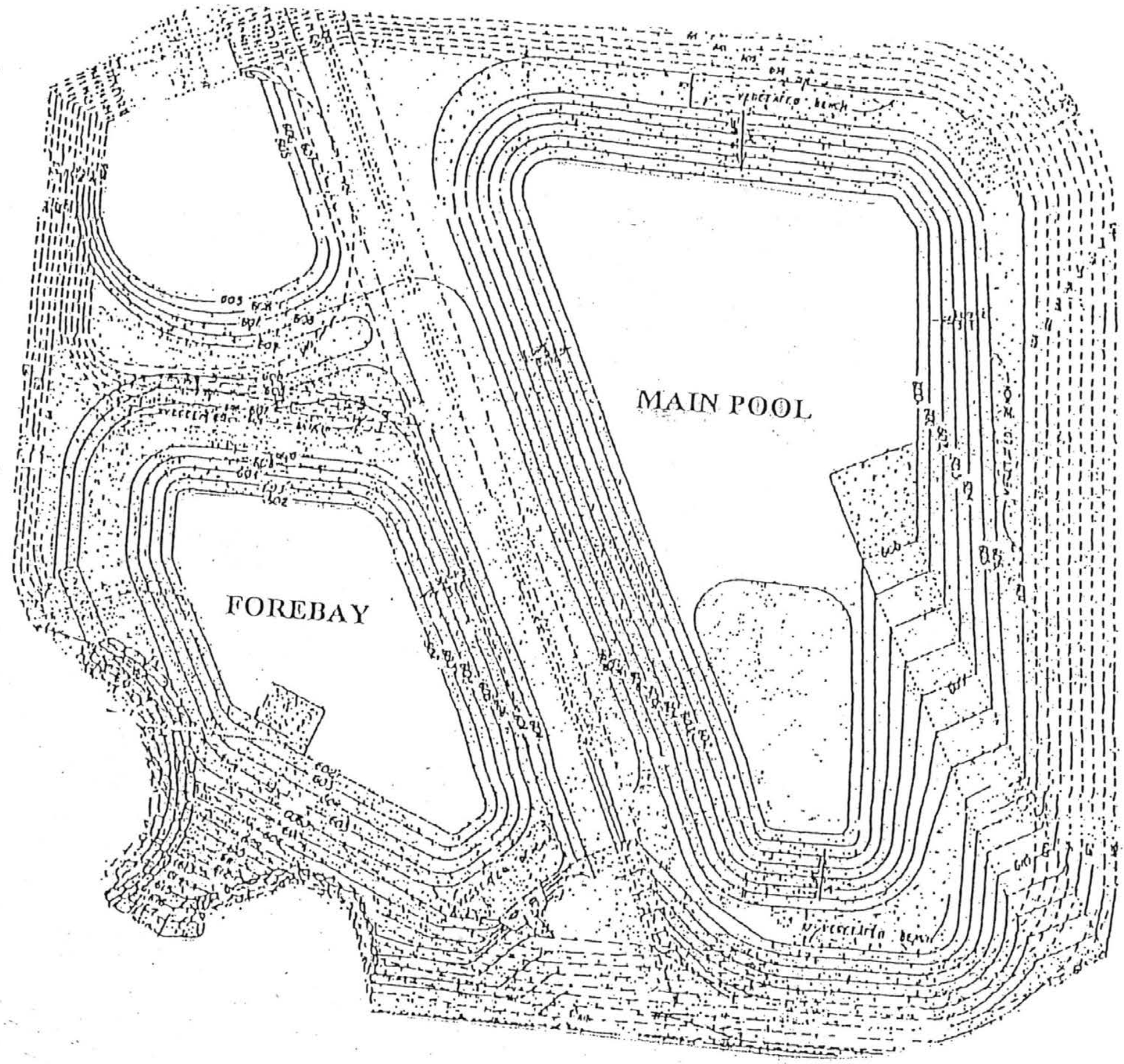


Fig. 9 Central Park mixed-use wetpond. Record drawings obtained from City of Austin Development Assistance Center.



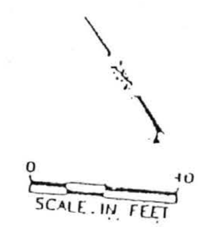


Fig. 10. Alpine mixed-use wetpond. Record drawings obtained from City of Austin Development Assistance Center.

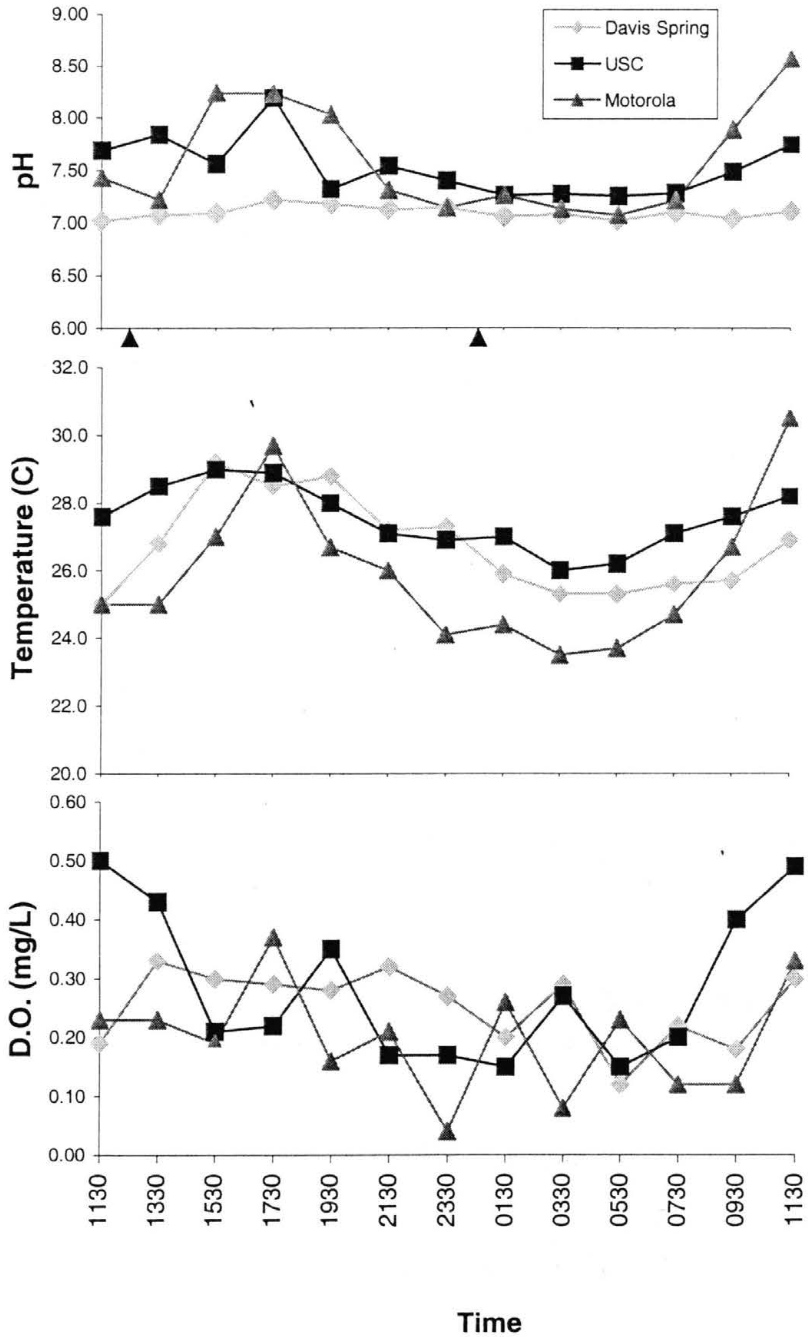
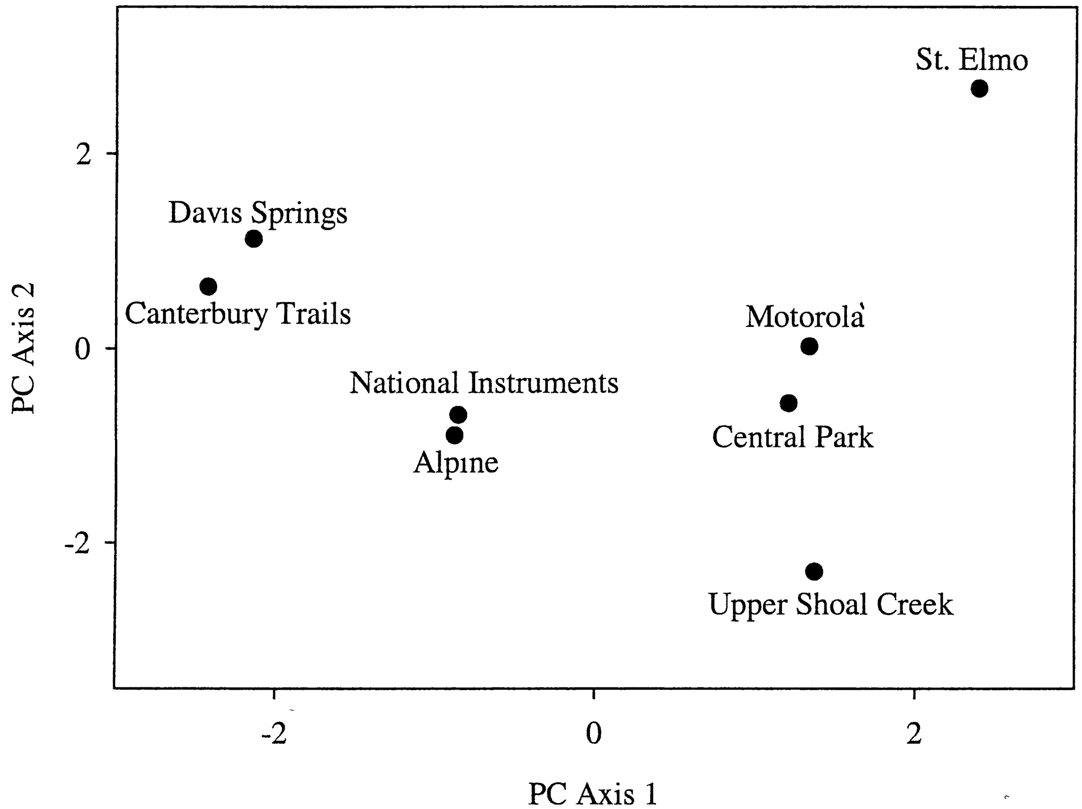


Fig. 11. Diel water pH, temperature (°C) and dissolved oxygen (mg/L) at Davis Spring, Upper Shoal Creek (USC) and Motorola wetponds on October 4-5, 2002.



	Axis 1 Eigenvector	Axis 2 Eigenvector
Age	0.48	0.09
Commercial use	0.24	0.22
Commerical/Residential use	0.15	-0.44
Impervious cover	0.50	0.01
Permanent pool volume	0.29	0.49
Pool surface area	0.34	-0.29
Residential use	-0.43	0.24
Surface area:volume	0.20	0.36
Watershed area	0.10	-0.48

Fig. 12. Scatter plot ordinating wetponds based on PC Axis 1 and PC Axis 2 variables.

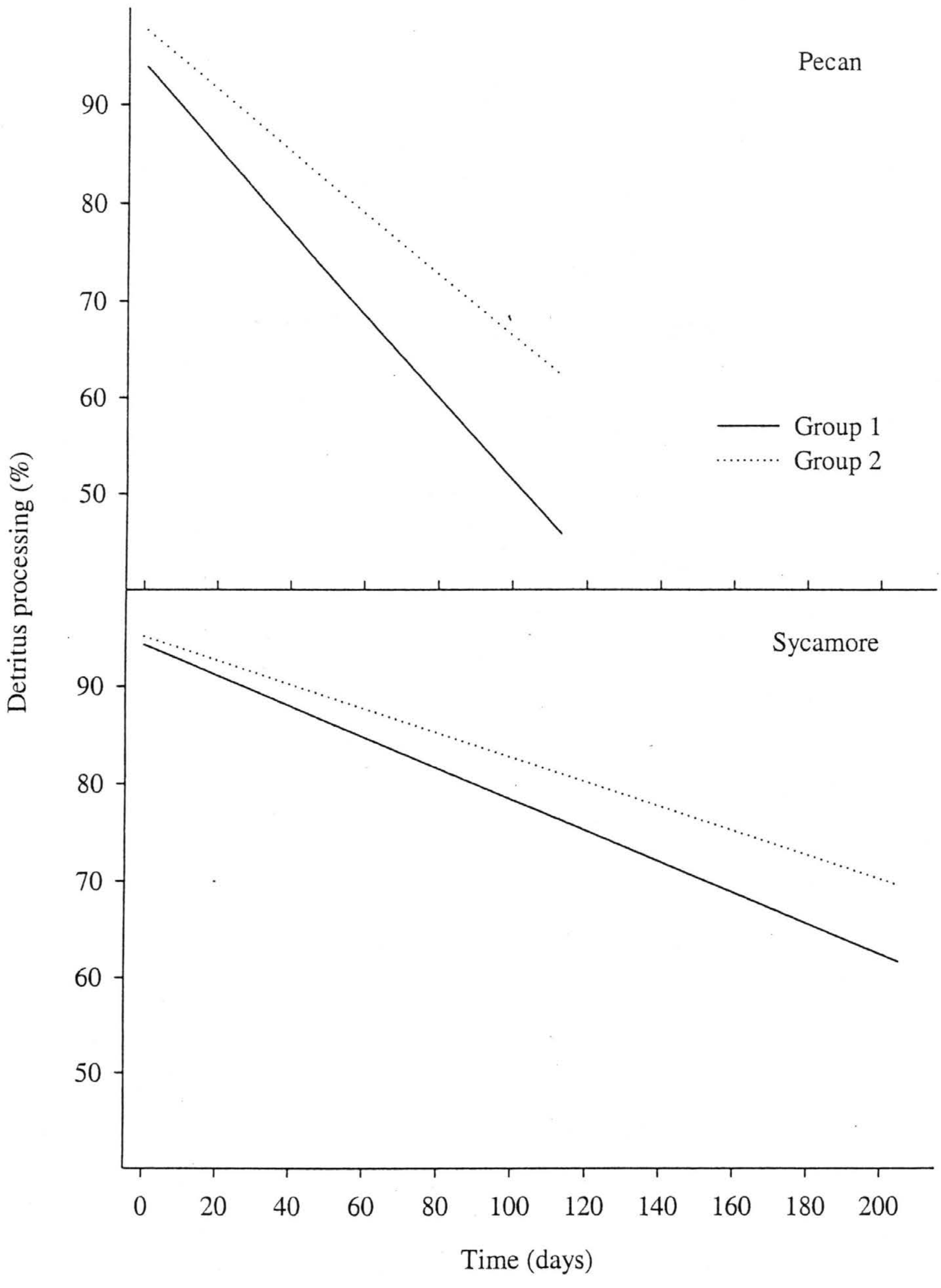


Fig. 13. Comparison of detritus processing slopes over time for PCA Group 1 and Group 2 wetponds for pecan and sycamore leaves. Group 1 = rapid decomposing wetponds. Group 2 = slow decomposing wetponds.

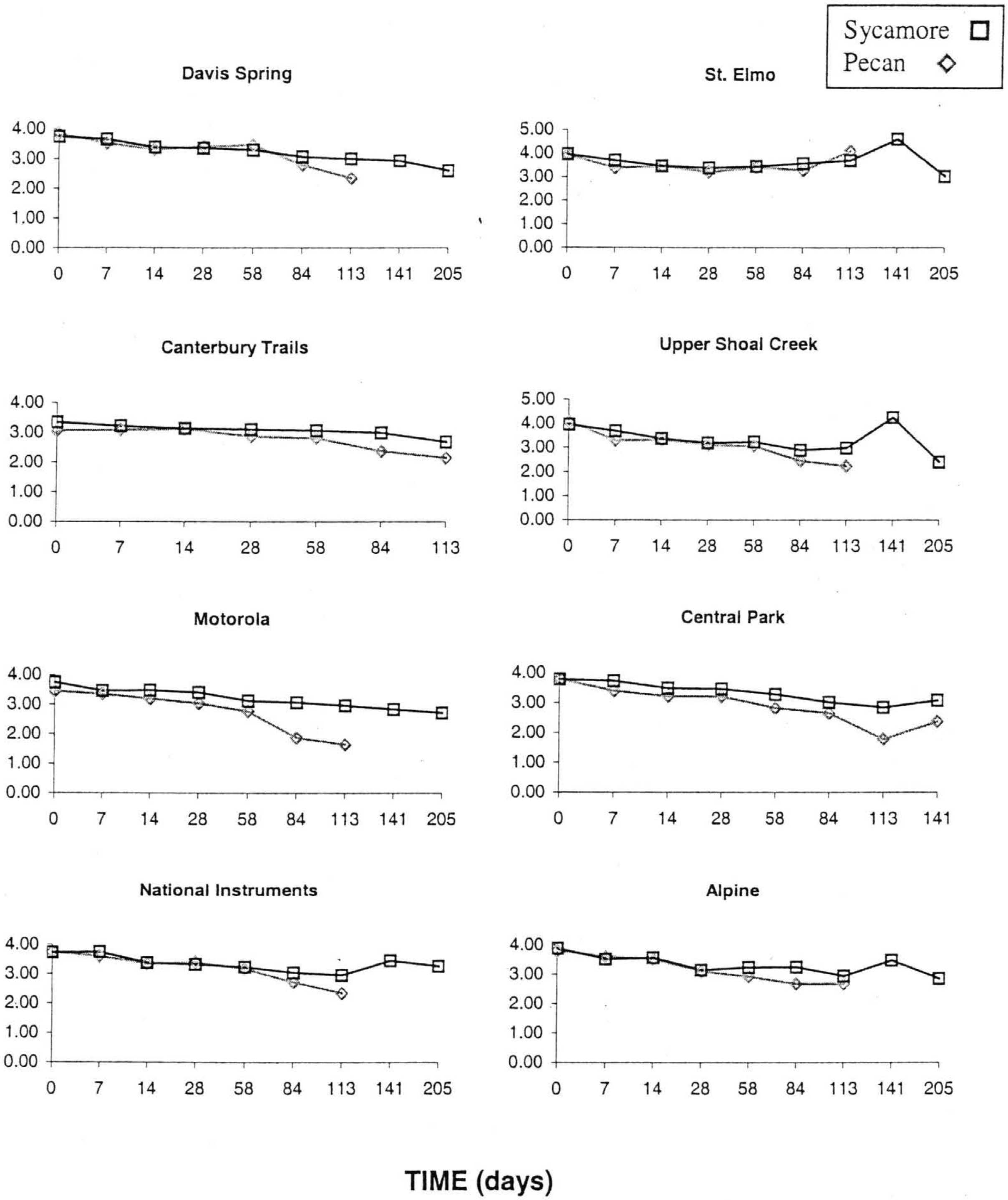


Figure 14. Leaf Particulate Dry Mass of Pecan and Sycamore over Time at Wetpond Sites

VITA

Kristine Marie Dennis was born in Charleston, South Carolina, on January 22, 1973, the daughter of Anthony and Jo Ann Dennis. After completing her work at Madison High School, San Antonio, Texas, in 1991, she entered Texas A&M University in College Station, Texas. She received the degree of Master of Science in Rangeland Ecology and Management in May 1996 During the following years she was an Americorps Volunteer for USFWS Santa Ana National Wildlife Refuge in Alamo, Texas, and for USFWS Ecological Services Field Office in Austin, Texas. Afterward, she was employed as a biologist for two years with Turner, Collie and Braden, Inc., in Austin, Texas. During the summer of 1999, she was employed as a Student Conservation Association Crew Leader in New Hampshire. In September 1999, she entered the Graduate School of Southwest Texas State University, San Marcos, Texas.

This thesis was typed by Kristine Marie Dennis.