EVALUATION OF NUTRIENT LEACHING AND NUTRIENT USE EFFICIENCY FROM MIXED SPECIES RESIDENTIAL LANDSCAPES

By

ZHIXUAN QIN

A THESIS PRESENTED TO THE GRADUATE SCHOOL
OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT
OF THE REQUIREMENTS FOR THE DEGREE OF
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To my beloved parents in China

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ABSTRACT OF THESIS PRESENTED TO THE GRADUATE SCHOOL
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Maintenance of turfgrass and ornamental plant cover in residential landscapes has the potential to negatively impact water quality due to frequent irrigation and fertilizer inputs. Studies have shown that areas planted with establishing woody ornamental plants had higher risk of nutrient leaching losses than turfgrass monoculture due to difference in root biomass. However, little information is available about nutrient leaching and nutrient use efficiency from established residential landscapes containing woody ornamental plants and turfgrass. The objectives of our study were to: 1) determine the effects of plant cover (turfgrass vs. woody ornamental) on nutrient leaching by quantifying nitrogen and phosphorus losses from simulated urban landscapes and 2) calculate the soil surface nutrient budgets for mixed landscape plots containing varying proportions of turfgrass and woody ornamental species.

Mixed landscapes consisting of three different turfgrass to woody ornamental proportions (60% turfgrass, 40% ornamental; 75% turfgrass, 25% ornamental; and 90% turfgrass, 10% ornamental) were installed in a completely randomized design with three replicates in nine drainage lysimeters. Turfgrass was fertilized following University of

Florida-Institute of Food and Agricultural Sciences (UF-IFAS) recommendations. Woody ornamental plants were fertilized by a production level rate to accelerate plant growth in three years. All of the species were irrigated based on their evapotranspiration and projected surface area of the plant canopy. Daily leachate samples were collected and combined to produce weekly flow-weighted samples for a period of one year. Leachate samples were analyzed for the concentrations of forms of N (total Kjeldahl N, nitrate-N (NO_3+NO_2–N), NH_4–N) and P (dissolved reactive P). The total mass losses (load) of N and P were calculated by multiplying the concentration by the total leaching volume. Cumulative loads of nutrients were determined by adding up nutrient loads over the total one year collecting period. Soil surface nutrient budgets (N and P) were calculated for each lysimeter by determining nutrient inputs (rainfall, irrigation, fertilizer), storage (Mehlich 3 soil P), and outputs (plant removal, nutrient leaching).

Our results indicated that leachate volumes were consistent with rainfall distribution with lower leachate volumes collected in dryer periods. There were many weeks (mainly in dry periods) that no plant cover treatment effect was noted on leachate volume, nutrients (N and P) concentrations and loads due to the fact that very low or even zero amount of leachate volumes was collected. In weeks with significant treatment effect, 90% turfgrass treatment leached higher amounts of leachate volume, and nutrient concentrations and loads (all tested forms) than 60 and/or 75% turfgrass treatment in most cases, suggesting improved nutrient use efficiency and reduced nutrient losses with the inclusion of more woody ornamental plants in mixed landscape. We suspected that the established woody ornamental plants with a significant increase of root biomass and density could fully colonize landscape plots and would be more

effective to absorb water and nutrients from soils compared with establishing plants. Annual nutrient budget calculation indicated N and P surpluses in our landscapes, where the nutrient (N and P) balances increased with the increasing proportion of woody ornamentals (60% turfgrass > 75% turfgrass > 90% turfgrass). This was mainly because woody ornamentals were over fertilized according to UF-IFAS recommendations in our study. However, even though woody ornamentals were fertilized at a production rates to accelerate growth, treatments containing a higher proportion of woody ornamentals still leached significantly less nutrient loads than treatments containing a higher proportion of turfgrass. The ability of woody ornamentals in landscape to remove nutrients may be even better if better fertilizer management practices were adopted (e.g., following fertilizer recommendations for landscape-grown ornamentals or split fertilization). As a result, we conclude that the use of woody ornamental plants in urban green areas improve nutrient use efficiency in the long term once roots fully expand allowing them to intercept water and nutrients.

CHAPTER 1
LITERATURE REVIEW AND RESEARCH OBJECTIVES

Nonpoint Source Pollution and Water Quality Concerns

Point-source pollution has been effectively controlled during the last several decades because it is easy to identify and place controls on these pollutant sources (USEPA, 2005). However, non-point source (NPS) pollution remains an urgent and serious problem because reduction of point source pollution has not resulted in significant improvements in water quality (USEPA, 2005). Non-point source pollution is more complex and harder to control than point-source pollution because NPS pollution is a diffuse source that occurs over large land areas and is subject to extreme temporal variability due to weather (Carpenter et al., 1998). Agricultural sources, like fertilizer and manures, are major contributors of NPS pollution (David and Gentry, 2000). Non-point source leaching and runoff nutrient losses (mainly nitrogen (N) and phosphorus (P)) waste precious plant nutrients and can harm natural ecosystems and human health.

Nitrogen, mainly in the form of nitrate (NO_3^-), is prone to movement in both surface and subsurface water flow because NO_3^- is soluble in water (Pathak et al., 2004) and is repelled from soil exchange sites due to its negative charge. Thus, NO_3^- generally is at high risk for loss in runoff or leachate (Mathers et al., 2007). Ammonium (NH_4^+) is another chemical form of N that is found in soils. Ammonium is positively charged and subject to reaction on the exchange complex on soil surface. Therefore, it is considered to be fairly immobile and less likely than NO_3^- to leach from most soils (Bowman et al., 2002). However, NH_4^+ can be quickly transformed to NO_3^- in the presence of oxygen (e.g., in well-drained soils) by the process of nitrification, and subsequently increase the chance of N loss from soils (Simek, 2000).

In contrast, P is more immobile than N due to the chemical bonding of P with metals and soil functional groups or precipitation of P minerals, especially in finer-textured soils or soils with high concentrations of Al, Fe, or Ca, (Borggaard et al., 2005; Tunesi et al., 1999). As a result, people used to regard surface runoff and sediment losses as the main pathways for P loss (mainly as particulate P) from most soils (Sims et al., 1998). However, researchers more recently identified P leaching (mainly as dissolved P) as an important mechanism of P loss from fertilized soils with low P sorption capacity (i.e., sandy soils with low concentrations of Al, Fe, or organic matter) (Breeuwsma and Silva, 1992; Hongthanat et al., 2011; Olson et al., 2010; Sallade and Sims, 1997; Schwab and Kulyingyong, 1989) or soils with high P levels resulting from repeated applications of P in fertilizers or manures (Heckrath et al., 1995). Studies have shown that P loss potential increases significantly when soil P concentrations reach a threshold or a change-point following repeated application of P fertilizers (Chakraborty et al., 2011; Maguire and Sims, 2002; Nair et al., 2004; Sims et al., 2002) For example, Sims et al. (2002) found that the concentration of dissolved reactive P (DRP) in runoff increased dramatically when soil Mehlich 3 P saturation ratio ($P_{M3}/(Al_{M3} + Fe_{M3})$), where P_{M3}, Al_{M3}, and Fe_{M3} is the concentration of Mehlich-3 extractable P, Al, and Fe expressed in mmol kg^{-1}) was higher than 0.14 for five Delaware soil series (Butlerstown, Evesboro, Matapeake, Pocomoke, and Sassafras). In another study, Maguire and Sims (2002) reported that DRP concentrations in leachate were low (below 0.1 mg P L^{-1}) until reached a Mehlich 3 P saturation ratio change-point of approximately 0.23, above which leachate DRP concentrations increased dramatically for the same five Delaware soils. In Florida sandy soils, Nair et al. (2004) showed that the changing point of degree of P

saturation ((M3-P)/α(M3-Fe+M3-Al))×100, where M3-P, M3-Fe and M3-Al is the concentration of Mehlich-3 extractable P, Al, and Fe expressed in mg kg^{-1}, α is an empirical factor considering P saturations of different soils) was 16%, above which water soluble P in soils increased significantly.

Nitrogen and P are both linked to accelerated eutrophication (Correll, 1998; Ryther and Dunstan, 1971; Sharpley et al., 2003). The delivery of excess N and/or P into water bodies stimulates the growth of algae or cyanobacterial mats. The high respiration rates of the excess cyanobacteria population and the decomposition of dead algae and aquatic weeds (e.g., alligator weed, water hyacinth, etc.) reduce the concentrations of dissolved oxygen and lead to the formation of anoxic layers within the water body (Sharpley et al., 2003) especially in waters with little movement and mixing effect (Correll, 1998). Fish and other aquatic species decline due to the depletion of oxygen within the water column. The overall biodiversity is reduced and water quality deteriorates, all of which directly impact the ability of a water body to function for its intended use (e.g., fisheries, recreation, drinking water, etc.) (Correll, 1998; Howarth; 1988). Researchers found that coastal water bodies are more vulnerable to NO_3–N pollution (Valiela et al., 1997); while PO_4–P is often regarded as the limiting nutrient in freshwater systems (e.g., lakes, streams, etc.) (Correll, 1998). Nitrogen and P concentrations as low as 0.05 to 0.1 mg L^{-1} NO_3–N (Burkholder et al., 1992) and 0.05 mg L^{-1} PO_4–P can result in water quality degradation(Mallin and Wheeler, 2000; Paerl et al., 1995). In addition to ecological consequences, excess nutrients in water bodies can also endanger human safety. Nitrate N concentrations above 10 mg L^{-1} in drinking water

are linked to human health issues including methemoglobinemia (in infants) and non-Hodgkin's lymphoma (Ward et al., 1996).

Urban Development and Nutrient Losses

Non-point source pollution of surface and groundwater from urban areas has become a major concern in recent decades due to the explosion of global population and expansion of urban areas (Fisher et al., 2006; Hauxwell et al., 2001) Researchers have linked water quality degradation to urban land use (Brett et al., 2005; Jordan et al., 2003), urban-based human activities (e.g., soil disturbance, fertilizers, septic system, etc.) (Fisher et al., 2006; Hauxwell et al., 2001), and runoff originated from new residential construction (Gregory et al., 2006; Line et al., 2002). Urbanization processes can alter the natural land cover, hydrology, and nutrient cycle and flows in urban system (Roach et al., 2008). The physical and chemical characteristics of urban soils are often very different from the natural soils that existed before development (Jim, 1998). Characteristics like alkaline soil pH, high bulk density (Gregory et al., 2006; Jim, 1998), poor soil structure, low organic content and soil fertility (Law et al., 2004), low nutrient and water holding capacity, and low biological activity were documented in urban soils. Frequent inputs of irrigation and fertilizer are often needed to achieve adequate plant quality and growth in urban green areas as a result of the low soil quality of many disturbed urban soils. Frequent applications of irrigation and fertilizers also increase the potential for nutrient losses from urban landscapes.

Fisher et al. (2006) modeled the effects of urbanization and loss of agriculture in the Choptank River basin in Maryland (USA) using the Generalized Watershed Loading Functions (GWLF) hydrochemical model. The authors showed that application of fertilizers and human wasterwater discharges were the major reasons of increased

nutrient export from Choptank River, which resulted in algal biomass production and increased turbidity in the estuary. The authors summarized that the intensity of urban land use, human population growth, and associated human activities (such as the application of fertilizers) can result in eutrophication even in areas where land cover is not changing significantly over time (Fisher et al., 2006). Similarly, Line et al. (2002) monitored runoff pollution from six drainage areas under various land uses (i.e, single-family residential, golf course, industrial, dairy cow pasture, construction site, and wooded) from June 1997 to April 1999. The authors reported that total N export was highest from the construction site, followed by single-family residential and the golf course. The greatest total P export was observed from the golf course, followed by pasture and residential land. The wooded site had the lowest total N and P export, with levels more in line with predevelopment nutrient exports (Line et al., 2002). These studies (Fisher et al., 2006; Line et al., 2002) showed that land use change associated with urban growth can greatly increase the potential risk for NPS nutrient losses. Therefore, it is meaningful to evaluate nutrient losses (mainly in leaching and runoff) from urban systems to understand the magnitude and pathway of nutrient losses and to identify possible best management practices (BMPs) to control NPS.

Nutrient Losses in Leachate and Runoff from Urban Landscapes

Nutrient Losses from Turfgrass Systems

Turfgrass is a typical type of vegetation planted in urban and residential landscapes across the U.S. It can serve as an effective nutrient filter because of its high-density biomass and thatch-forming capabilities, which help to slow runoff velocities, reduce sediment loss, and increase infiltration (Linde et al., 1995, 1998; Petrovic, 1990). However, nutrient losses from turfgrass systems can still occur due to

intense precipitation (Snyder et al., 1984), saturated soil (Baird et al., 2000), soil disturbance (Barton and Colmer, 2006), fertilizer source (Brown et al., 1977), and irrigation strategies (Snyder et al., 1984). Many researchers evaluated nutrient losses from turfgrass (Brown et al., 1977, 1982; Morton et al., 1988; Mosdell and Schmidt, 1985; Nelson et al., 1980; Petrovic, 1990; Petrovic et al., 1986; Sheard et al., 1985; Snyder et al., 1981, 1984). In a review of the literature, Petrovic (1990) reported N leaching ranged from 0 to 80% of applied N fertilizer to turfgrass depending on factors including soil type, N source and rate, timing of fertilizer application, and use of irrigation.

Soil texture was reported as a factor affecting nutrient leaching from turfgrass systems (Petrovic, 1990). Petrovic (2004) evaluated soil texture effects on N leaching from creeping bentgrass (*Agrostis stolonifera* L.) fertilized with methylene urea in a field experiment. The results showed that 9.1, 1.5, and 3.5% of the applied N was leached from a sand soil, sandy loam soil, and silt loam soil, respectively (Petrovic, 2004). In another study, Soldat and Petrovic (2008) summarized the results of eight studies that measured P leaching from both cold-season and warm-season turfgrasses grown on different soil textures (sand, loamy sand, sandy loam, silt loam, sandy clay, and clay) in field or greenhouse experiments. They reported annual P leaching losses from fertilized turfgrass grown on sandy soils ranged from 0.03 to 6.1 kg ha^{-1} (18.5 kg ha^{-1} in a greenhouse study), with the highest observed P concentrations exceeding 13 mg L^{-1}. In contrast, P leaching from fertilized turfgrass grown on fine-textured soils ranged from 0.2 to 5.4 kg ha^{-1} (Soldat and Petrovic, 2008). Results of these studies (Petrovic, 2004; Soldat and Petrovic, 2008) suggest that careful attention must be paid to soils with

coarser texture to prevent high potential of NPS nutrient losses from areas planted with turfgrass.

Fertilizer source and rate were also reported as factors influencing the magnitude of nutrient leaching from turfgrass. Shuman (2003) determined the effects of fertilizer sources on N and P leaching from simulated golf greens in a greenhouse study. The lowest cumulative leachate mass load of NO_3–N was observed when simulated greens were fertilized with sulfur-coated urea (0.7% applied N) or polymer- and sulfur-coated micro-granular (13-13-13) (1.4% applied N) (Shuman, 2003). The other three treatments (soluble 20-20-20, agriculture grade granular 10-10-10, and liquid controlled-release N with super phosphate) leached comparable amounts of cumulative NO_3–N of 3.3 to 6.4% applied total N when all five fertilizers were applied at 24 kg ha^{-1} N. However, the results indicated that leachate from golf greens receiving soluble fertilizers (20-20-20 and 16-25-12) had the highest cumulative leached PO_4–P mass (both 43% of applied P). The amount of PO_4–P leached from golf greens receiving other fertilizer sources (polymer- and sulfur-coated micro-granular 13-13-13, agricultural granular 10-10-10, soluble (19-25-5 and 9-18-18), sulfur-coated urea with super phosphate and liquid controlled-release N with super phosphate) varied from 15 to 25% of total P applied when all of the eight fertilizers were applied at a rate of 11 kg ha^{-1} P (Shuman, 2003). The author summarized that coated, controlled release granular fertilizer were effective at reducing NO_3–N leaching losses, but did not affect PO_4–P leaching mass compared with other granular source (Shuman, 2003). Easton and Petrovic (2004) evaluated the effect of nutrient source (i.e., dairy and swine compost, biosolids, soluble urea, and sulfur-coated urea) on N and P in leachate from establishing turfgrass (Kentucky bluegrass and

perennial ryegrass) planted in a sandy loam soil in a mass-balance field study. Fertilizers were applied to plots at two application rates (50 and 100 kg ha^{-1} N) to achieve a total annual application of 200 kg ha^{-1} N. Phosphorus application rate varied depending on the N application rate and the P content of the fertilizer source. Significantly higher NO_3–N leaching losses in average were reported from plots fertilized with synthetic organic fertilizers (soluble urea: 186.5 kg ha^{-1}; sulfur-coated urea: 181.7 kg ha^{-1}) than natural organic fertilizers (biosolid: 132.3 kg ha^{-1}; swine compost: 48.7 kg ha^{-1}; dairy compost: 44 kg ha^{-1}) at the 100 kg ha^{-1} N application rate. However, two of the natural organic fertilizers leached significantly higher PO_4–P mass (swine compost: 10.6 kg ha^{-1}; dairy compost: 10.0 kg ha^{-1}) in average than all synthetic organic fertilizers (soluble urea: 2.2 kg ha^{-1}; sulfur-coated urea: 1.7 kg ha^{-1}) at the same 100 kg ha^{-1} N application rate. The relatively high P losses from the natural organic sources were due to the higher P application rate (average P application rate of natural organic and synthetic organic were 39.4 and 14.8 kg ha^{-1}) when fertilizers were applied based on an N basis (100 kg ha^{-1} N) (Easton and Petrovic, 2004). The authors also reported significantly lower masses of N and P were leached from turfgrass plots for all fertilizer treatments when fertilizers were applied to the plots four times at 50 kg ha^{-1} N rate instead of two times at 100 kg ha^{-1} N, suggesting that reducing application rate per fertilization event can reduce nutrient losses (Easton and Petrovic, 2004).

The timing of fertilizer application also affects leachate nutrient losses from turfgrass. Erickson et al. (2010) reported that delaying the fertilization of sod until 30 d after installation significantly reduced leachate NO_3–N and PO_4–P losses when compared with fertilization immediately after the day of sod installation. The authors

reported that it was unnecessary to apply fertilizer to sod at installation because it can result in significant NO_3–N and PO_4–P leaching (Erickson et al., 2010). Other studies also showed that establishing turfgrass systems have a higher risk of nutrient losses than established turfgrass. Miltner et al. (1996) used ^{15}N-labeled urea to track the leaching and mass balance of N input to Kentucky bluegrass turf grown on a sandy loam soil. The authors reported significant higher loads of ^{15}N leached (4.05 kg ha^{-1} yr^{-1} ^{15}N) from 6 month old turfgrass than from established sod (2 years) (1.65 kg ha^{-1} yr^{-1} ^{15}N). Fertilizer is in need for rapid establishment and growth of turfgrass (Easton and Petrovic, 2004); however, less fertilizer should be applied during turfgrass establishment to minimize nutrient losses during unreactive root-uptake time (i.e., reduced uptake ability due to establishing rooting system). Studies also suggested that fertilizer application should be avoided before abundant precipitation event since nutrients applied can be subject to losses in runoff or leachate (Brown et al., 1977; Engelsjord and Singh, 1997; Erickson et al., 2010; Miltner et al., 1996; Morton et al., 1988; Snyder et al., 1984; Snyder et al., 1981). For example, Erickson et al. (2010) reported that higher drainage was collected from fertilized sod growing in wet season (740 mm) than fertilized sod growing in dry season (238 mm), which subsequently increased the risk of dissolved nutrients losses. In another study, Snyder et al. (1984) also reported particularly higher total N (NH_4 + NO_3–N) leaching (40.5 to 53.6% of applied N) during first fertilization cycle (February-March) from areas planted with Bermudagrass due to the intense precipitation event after fertilization during this period (154 mm of rain during the first 10 days following NH_4NO_3 application).

Irrigation also affects nutrient leaching from turfgrass. Morton et al. (1988) reported that overwatering (3.75 cm wk^{-1} in addition to rainfall) treatment had significantly higher NO_3 mass loss than scheduled irrigation treatment (1.25 cm once the measured soil-water potential was -0.05-MPa tension) at two fertilizer rates (97 and 244 kg N ha^{-1} yr^{-1}) from a Merrimac sandy loam planted with Kentucky bluegrass in Kingston, RI. The author summarized that overwatering in conjunction with fertilization (especially high fertilization rate) generated high risk of nutrient losses (Morton et al., 1988). Similarly, Barton and Colmer (2006) summarized that low N losses (<5% of applied fertilizer N) tended to leach from established turfgrass that was not over-irrigated (i.e., irrigation at a rate that equal to or less than potential evapotranspiration) but received N fertilizer at 200 to 300 kg ha^{-1} yr^{-1}. In another study, Starrett et al. (1995) investigated N leaching from undisturbed soil columns (20 cm in diameter and 50 cm in depth) planted with a Kentucky bluegrass under two irrigation regimes (heavy: four 2.54 cm applications, and light: sixteen 0.64 cm applications) by tracing applied ^{15}N-labeled urea. The results indicated that heavy irrigation increased mean ^{15}N collected in leachate by 30 times compared with soil columns receiving light irrigation. The authors concluded that N leaching can be reduced by replacing infrequent irrigation at high rate with frequent irrigation at low rate when the same amount of irrigation was applied (Starrett et al., 1995). Pathan et al. (2007) compared a soil moisture sensor-controlled irrigation system with conventional irrigation scheduling (as recommended for domestic lawns) in Perth, Australia. The authors showed that the sensor-controlled system applied 25% less cumulative volume of water to turfgrass plots compared with watered based on a conventional irrigation schedule. Also, the use of sensor-controlled irrigation system

resulted in lower leachate volume (4% of applied irrigation plus rainfall) and mineral N (NO_3 + NH_4) in leachate (0.22 kg ha^{-1}) than a conventional irrigation system (16% of applied irrigation plus rainfall of leachate volume and 0.83 kg ha^{-1} mineral N) (Pathan et al., 2007). The results of these studies (Barton and Colmer, 2006; Morton et al., 1988; Pathan et al., 2007; Starrett et al., 1995) demonstrated that scheduling irrigation to match plant water needs prevents water and nutrients from leaching below the active rooting zone.

Turfgrass species also affects nutrient leaching from soils. Bowman et al. (2002) compared leachate N losses from six warm-season turfgrasses including common bermudagrass, 'Tifway' hybrid bermudagrass (*C. dactylon* ×*transvaalensis*), centipedegrass (*Eremochloa ophiuroides* (Munro) Hack.), 'Raleigh' St. Augustinegrass (*Stenotaphrum secundatum* (Walter) Kuntze), 'Meyer' zoysiagrass (*Zoysia japonica* Stead.), and 'Emerald' zoysiagrass (Z. *japonica* ×*tenulfolia*). Significant differences among species were reported for cumulative NO_3–N leached. St. Augustinegrass was noted as the most effective species for reducing leaching due to the high root length density (i.e., the length of roots per volume of soil) with just 0.9% applied N leached; ‘Meyer’ zoysiagrass was the least effective species with 17.6% of applied N leached. The rest of the species leached 7.3 to 9.3% of applied N. The authors suggested that the documented difference in NO_3–N leaching potential among species was related to root distribution. Thus, selection of turfgrass species should be considered as an important factor in urban turfgrass management to minimize NPS nutrient losses (Bowman et al., 2002).

Moreover, many studies indicated that the maintenance of healthy turfgrass, which has a dense cover and expansive root system, can effectively absorb nutrients when fertilizers are applied at recommended rates, thus minimizing leaching from landscapes (Brown et al., 1977; Easton and Petrovic, 2004; Frank, 2008). Hence, factors that affect the health of turfgrass affect the potential for nutrient losses from turfgrass systems (Petrovic, 1990). For example, Jiang et al. (2000) evaluated N losses following the sudden death of twelve-year-old turfgrass plots (all killed by nonselective herbicide glyphosate) that were comprised of four cool-season turfgrass species grown in an Enfield silt loam (coarse-silty over sandy or sandy-skeletal, mixed, active, mesic Typic Drystrudept) in Kingston, RI. The researchers showed that plots containing dead turfgrass leached significantly more cumulative NO_3–N (161 kg ha^{-1}) than healthy turfgrass plots (50 kg ha^{-1}) over a 12 month period (Jiang et al., 2000). In another study, Trenholm et al. (2012) recorded high losses of leaching NO_3–N load in spring and early summer fertilizer cycle (74 to 84% of annual NO_3–N load leached) from areas planted with zoysiagrass due to the presence of large patch disease (*Rhizoctonia solani* J.G. Kuhn) that resulted from high N application rate (343 and 490 kg ha^{-1}) that led to a reduction of live green tissue. The authors summarized that lower N application rate (≤ 196 kg ha^{-1}) should be applied to zoysiagrass to prevent or reduce plant disease, enhance turfgrass cover, and subsequently reduce nutrient loss (Trenholm et al., 2012). Therefore, maintenance of a vigorous turfgrass cover is important to reducing potential NO_3 contamination of ground water.

Nutrient losses in runoff from turfgrass systems are rarely reported in the literature. Gross et al. (1990) concluded that less N was lost in runoff from turfgrass (0.14 kg ha^{-1}

N and 0.02 kg ha^{-1} P) than from agronomic row crops (e.g., 11.7 kg ha^{-1} N and 2.4 kg ha^{-1} P from cultivated tobacco (Angle, 1985)) at the same study site. Petrovic (1990) recorded a study which observed a single runoff event (following a natural precipitation event) from an established turfgrass planted in silt loam soil with high slope (9 to 12%). The high organic inputs to soils under turfgrass by litter and root improve the soil structure, and hence increase water infiltration capacity of the soils, which reduces the potential for nutrient runoff events occurring in urban landscapes (Petrovic, 1990). Similarly, Morton et al. (1988) only observed two surface runoff events from turfgrass plots growing on a Merrimac sandy loam. Both runoff events resulted from unusual climatic conditions (one generated by rainfall on frozen ground, the other followed a big storm with 12.5 cm precipitation in one week) during a 2 yr study. However, nutrient losses during runoff events can still occur from turfgrass planted on fine-textured soils or high bulk density soils (compacted conditions), or during intense precipitation events (Hipp et al., 1993; Moe et al., 1967). Also, the water holding capacity of soils and soil moisture status before a rainfall or irrigation event are important factors affecting runoff nutrient losses from turfgrass. Linde and Watschke (1997) showed off-site nutrients movement when runoff occurred shortly after granular fertilizer application to a nearly saturated soil on golf fairways. The authors suggested the importance of irrigation management to ensure that soils are not maintained under saturated conditions (Linde and Watschke, 1997). In another study, Shuman (2002) evaluated nutrient losses in runoff following fertilizer application to turfgrass. The results showed that the runoff volume was determined by precipitation amounts and soil moisture condition before precipitation, with the highest concentrations and mass of PO_4–P in runoff noted when

simulated rainfall begun 4 h after a fertilization; concentrations and loads of PO_4–P in runoff decreased dramatically when simulated rainfall was applied 24 h after the fertilization. The author suggested minimal amounts of irrigation should be applied after fertilizer application and fertilizer applications before intense rain or during high soil moisture condition should be avoided to minimize the potential for nutrient losses in runoff from turfgrass (Shuman, 2002).

Nutrient Losses from Ornamental Landscape Plant Beds

Ornamental plants (e.g., trees, shrubs, perennials, annuals, etc.) are another type of vegetation commonly found in urban landscapes. However, few research studies about nutrient losses from ornamental plant beds are available in the literature compared with research on turfgrass nutrient losses. Factors like fertilizer type (Altland et al., 2002; Marble et al., 2011) and vegetation type (Amador et al., 2007; Hipp et al., 1993) have reported to influence nutrient losses from ornamental plant beds. As was reported for turfgrass systems, fertilizer type affects nutrient losses from ornamental landscapes. Marble et al. (2011) compared nutrient leaching from bedding plants fertilized with composted poultry litter (CPL) or commercial inorganic fertilizer (Peafowl brand garden-grade fertilizer (13-13-13) and Polyon (13-13-13)). The researchers showed that the use of CPL in landscape beds resulted in less N leached than when commercial inorganic fertilizer was applied to beds at the same N application rate (e.g., 0.7 and 44.7 mg L^{-1} NH_4–N leached from CPL and inorganic fertilizer treatment four weeks after planting, respectively when applied at the 9.8 g m^{-2} N application rate). The authors stated that the use of CPL may reduce the required inorganic fertilizer inputs and provide a more environmentally-friendly alternative for the growth of annual beds (Marble et al., 2011). In another study, Altland et al. (2002) reported less N (NO_3–

$N+NH_4$–N) leaching below the plant roots of annual ornamental flowers when beds were fertilized with controlled release fertilizer (8.1 and 5.3 mg L^{-1} in 2 and 4 weeks after planting respectively) compared with beds fertilized with granular water soluble fertilizers (9.5 and 9.1 mg L^{-1} in 2 and 4 weeks after planting respectively) at the same N application rate. Results of these studies (Altland et al., 2002; Marble et al., 2011) indicate that the use of lower solubility fertilizers (e.g., controlled release fertilizer, organic fertilizers) can help reduce nutrient leaching from ornamental landscapes.

Other studies showed that vegetation type in ornamental landscapes also influences nutrient loss potential. Amador et al. (2007) compared N losses from unplanted mulched areas, areas planted with ground covers, turfgrass, deciduous trees, evergreen trees, evergreen shrubs, deciduous shrubs, annual flowers, perennial flowers, and a native woodland. The authors showed that unplanted mulched areas and areas planted with ground covers leached highest amount of NO_3–N and acted as a source of NO_3–N to groundwater (> 10 kg ha^{-1} N), followed by deciduous and evergreen trees, and turfgrass (2 to 10 kg ha^{-1}). The rest of the vegetation types were very effective at reducing NO_3–N losses (< 2 kg ha^{-1}). The authors concluded that priority should be given to native woodlands, annual and perennial flowers, and deciduous and evergreen shrubs when designing and managing sustainable landscapes to protect groundwater quality (Amador et al., 2007). Hipp et al. (1993) reported that native or resource-efficient plants, such as salvia (*Salvia officinalis* L.), wax myrtle (*Morella cerifera* (L.) Small), and yaupon holly (*Ilex vomitoria* Sol. ex Aiton), required low water and chemical inputs since they were well adapted to the Dallas, TX (USA) study area, and subsequently resulted in lower nutrient losses potential than high-maintenance landscapes. The authors

suggested that the installation of resource-efficient plants in urban areas “could provide cities and suburban areas with a cost-effective, low-maintenance, and aesthetically-pleasing pollution control technology” (Hipp et al., 1993).

Nutrient Losses from Mixed Species Landscapes

The typical urban and residential landscape in the U.S. consists of a mixture of turfgrass and ornamental plants (e.g., trees and woody shrub species) (Hipp et al., 1993). Recently, researchers began investigating how vegetation type (turfgrass vs. ornamental plants) influences nutrient leaching from urban landscapes. Results of several studies supported the hypothesis that establishing woody ornamental plants systems have a higher nutrient loss risk potential than turfgrass (Cisar et al., 2004; Erickson et al., 2001, 2005; Loper et al., 2012). For example, Erickson et al. (2001) compared N losses (NO_3–N and NH_4–N) in leachate from newly established St. Augustinegrass and mixed ornamental landscape plots (planted with 12 ornamental species including perennial grasses, woody shrubs, and trees) in south Florida. The authors reported N loads in leachate were significantly greater from the mixed-species landscape (48.3 kg ha^{-1} N) during three fertilizer cycles than from St. Augustinegrass monoculture (4.13 kg ha^{-1} N). Cisar et al. (2004) reported that more N (NO_3–N + NH_4–N) was leached from mixed ornamental landscape plots (mainly woody ornamental plants) (annual mean leached N mass of 4.89 g m^{-2} and 32.6% of applied N fertilizer) than from turfgrass (annul mean leached N mass of 0.41 g m^{-2} and 1.4% of applied N fertilizer) even though woody ornamental plants require less fertilizer inputs than turfgrass (St. Augustinegrass). The authors summarized that St. Augustinegrass had higher N use efficiency than woody ornamental plants when properly maintained and suggested St. Augustinegrass as a better landscape type to reduce N loss to ground water (Cisar et

al., 2004). Erickson et al. (2005) also reported that leaching of P and K was higher from mixed species landscape plots (37.8 and 346 kg ha^{-1}, respectively) than from turfgrass monoculture plots (22.9 and 185 kg ha^{-1}, respectively). Similarly, Loper et al. (2012) reported higher nutrient (NH_4–N, NO_3 + NO_2–N, and DRP) leaching from newly planted woody ornamental plant beds than from turfgrass areas in the landscape. Results of these studies by Cisar et al. 2004, Erickson et al. (2001, 2005), and Loper et al. (2012), indicated that nutrient leaching may be enhanced when woody ornamental species are installed in place of turfgrass.

Results of most of the nutrient leaching studies conducted in mixed landscapes suggested that newly planted turfgrass sod had a higher root density and root biomass than newly planted woody ornamental plants (Cisar et al., 2004; Erickson et al., 2001, 2005; Loper et al., 2012). This difference in root density is suggested as the reason that nutrient losses in leachate were higher in woody ornamental plant beds than turfgrass monoculture (Cisar et al., 2004; Erickson et al., 2001, 2005; Loper et al., 2012). However, none of these mixed landscape leaching studies (Cisar et al., 2004; Erickson et al., 2001, 2005; Loper et al., 2012) were continued for a significant period past establishment in order to compare vegetation effects on nutrient losses from established landscapes. Erickson et al. (2008) reported significantly higher inorganic-N (NO_3–N and NH_4–N) loads in leachate from mixed species landscape (5.2 kg ha^{-1}) than from turfgrass monoculture (1.3 kg ha^{-1}) during year one of a 3-years study. However, inorganic–N leaching was not significantly different between the mixed-species and turfgrass landscapes in subsequent years. The authors attributed the higher nutrient loss condition from mixed species landscape in year one to the longer establishment

period for woody ornamental plants when compared with turfgrass sod, which was supported by the fact that root biomass of turfgrass monoculture was consistent during the study, while root biomass of mixed-species landscape increased from 100g m^{-2} at the beginning to 300 g m^{-2} after the first year. Thus, the fading of vegetation effect on nutrient losses may be explained by the significantly increase of root biomass of woody ornamental species once plants are fully established (Erickson et al., 2008). Moreover, most studies about nutrient losses from mixed landscape systems focused primarily on quantifying N leaching losses (Altland et al., 2002; Cisar et al., 2004; Erickson et al., 2001; Marble et al., 2011); only a few studies evaluated P losses in leachate (Erickson et al., 2005; Loper et al., 2012). Thus, there is a need to quantify nutrient losses (both N and P) from mature landscapes containing turfgrass and woody ornamental plants in order to develop appropriate BMPs to minimize NPS pollution from urban landscapes.

Nutrient Budgets as an Urban Nutrient Management Tool

Nutrient budgets are a tool that have been widely used to understand nutrient flows, identify nutrient surplus areas, and guide nutrient management planning in agriculture. In general, a nutrient budget analysis compares nutrient inputs (fertilizers, manures, biological N_2 fixation, etc.) with nutrient outputs (crop harvest etc.) within a specifically defined geographic area (Oenema et al., 2003; Slaton et al., 2004; Watson et al., 2002). A positive nutrient budget (inputs > outputs) suggests that nutrients have the potential to accumulate in nutrient sinks (e.g., soil, water bodies, biomass etc.) and indicates an increased risk of nutrient loss. In contrast, a negative nutrient budget (inputs < outputs) suggests that the nutrients in storage will be gradually depleted. For soil nutrient storages, a negative nutrient budget suggests that nutrients stored in soils may not be available for optimum plant growth (Oenema et al., 2003). Avoiding nutrient

surpluses is equally as important as preventing nutrient deficits because it reduces the chance that surplus nutrients move into water or air. Avoiding nutrient deficits ensures that crops or ornamental plants have sufficient levels of nutrients (mainly N and P) that are available for plant uptake. Nutrient balance (inputs = outputs) is the most ideal situation for agricultural and urban landscape management.

Oenema et al. (2003) and Watson et al. (2002) summarized three basic methods to quantify the nutrient budget. The farm-gate nutrient budget (black-box approach) records controlled nutrients that enter (e.g., fertilizers or manures) and leave (e.g., plant harvest or agricultural products) the farm system. The farm-gate budget approach can be used in any unit boundary and is not limited to farm units. The main source of data used in farm-gate nutrient budget is readily available annual statistical information and inventories. Uncontrollable inputs like biological fixation of N and atmospheric deposition are not considered in the farm-gate method (Watson et al., 2002). The farm-gate budget has been mainly used in policy analysis (Watson et al., 2002). A soil surface budget accounts for all nutrients that enter and leave (crop and/or animal uptake) the soil surface. A soil surface budget takes into consideration of possible nutrient losses and inputs by estimating volatilization (N that does not enter the soil), atmosphere deposition, and biological N fixation of nutrients. But it ignores nutrient cycles and transformations (e.g., mineralization and immobilization) within the soil systems (Watson and Atkinson, 1999). The soil surface budget is used more often in agricultural system at field scale to determine the nutrient requirements of specific crop (Ministry of Agriculture Fisheries and Food, 2000; Watson et al., 2002). Finally, a soil system budget identifies all nutrient inputs and outputs by including nutrients

transformation and flows within the soil system, and nutrients gains and losses through the soil surface. The system approach identifies and estimates all possible nutrient fates in soil system (i.e., volatilization, denitrification, leaching loss, runoff, etc.). Besides statistical data, the data in soil system budget also include estimates (i.e., biological N fixation, denitrification), assumptions, and continuous precise measurements in the field and laboratory (e.g., leaching and runoff losses) to identify all possible components of the nutrient budgets (Oenema et al., 2003; Watson and Atkinson, 1999). However, the development of computer models reduces the need for field and laboratory measurements (Watson et al., 2002). The soil system budget aims at identifying the cause and fate of nutrient surpluses (Oenema et al., 2003).

There is no one correct nutrient budget approach to general nutrient budget calculations (Oenema and Heinen, 1999; Watson et al., 2002). The selection of a nutrient budget approach depends on the purpose of the budget calculation and the targeted nutrients types (Oenema and Heinen, 1999). Oenema et al. (2003) compared the three nutrient budget methods on an experimental dairy farm system in Netherlands. The ratio of unidentified N (i.e., total accounted N inputs minus N in all specific identified output pathways) to total accounted N input was 0.65, 0.33, and -0.01 (negative number suggesting outputs are higher than inputs) for the farm-gate, soil surface, and soil system budget approach, respectively. As a result, the soil system budget was the most detailed and complex nutrient budget calculation because it partitioned nutrient surpluses (unaccounted pathways) calculated by the farm-gate and soil surface budget by specific fate pathways (e.g., leaching, runoff, denitrification, etc.) and identified possible nutrient losses to environment (Oenema et al., 2003).

Nutrient budgets have been calculated at several different geographical scales ranging from farms (Buciene et al., 2003), watersheds (Watson et al., 1981), states (Lanyon et al., 2006; Slaton et al., 2004), regions (Sacco et al., 2003), and countries (Lord et al., 2002) to identify areas with nutrient surpluses. For example, Sacco et al. (2003) calculated soil surface nutrient budgets for the main farm types (cattle, dairy, traditional livestock, pig, and non-livestock) within a 7,773 ha area located in Northwest Italy. Nutrient surpluses were greatest for dairy farms (320, 110, and 320 kg ha^{-1} for N, P, and K, respectively), followed by pig livestock farms (270, 100, and 320 kg ha^{-1} for N, P, and K, respectively); cattle livestock farms had the lowest nutrient surplus (190, 80, and 230 kg ha^{-1} for N, P, and K, respectively). The authors suggested that manure exchange between farms in the area could be a practical way to reduce the use of mineral fertilizers to achieve acceptable N and K surplus level. However, P surplus within the area would not be ameliorated by this way (Sacco et al., 2003). By calculating a nutrient budget, the authors showed that it was important to regulate the use of solid manure to reduce nutrient surplus conditions (Sacco et al., 2003).

Watson et al. (2002) collated and summarized 88 nutrient budgets calculated at different farms (farm types including dairy, arable, poultry, horticulture, beef, sheep, and mixed) in nine temperate countries (Austria, Canada, Germany, Netherlands, New Zealand, Norway, Sweden, and UK). The results indicated an annual mean N surplus of 83.2 kg ha^{-1} among different farms, with the highest (90%) and lowest (20%) N use efficiencies (i.e., outputs/inputs) in arable (e.g., corn, soybeans, wheat, cereal, and fodder) and beef systems, respectively. Annual P budgets reported the greatest surplus for horticultural systems (vegetables and fruits) (average 122 kg ha^{-1}), which resulted

from the land-application of purchased manure. The data illustrated that the nutrient use efficiencies and nutrient balance conditions (i.e., surplus or deficit) varied with farm and nutrient types, and highlighted the importance of balancing N and P in farming system (Watson et al., 2002).

Obour et al. (2011) calculated a soil surface P budget for the Bh horizon of a Spodosol in Florida that was planted with bahiagrass and fertilized at three P application rates (0, 5, and 10 kg ha^{-1} P). The results showed that the net P balance changed from negative to positive for all treatments when considering the top 45 cm P storage in soils instead of P storage within 15 cm soils in budget calculation, suggesting that P in the Bh horizon was available and provided significant amount of P for pastures growing on Spodosols in Florida. The authors suggested that fertilizer inputs to perennial grasses grown should be reduced when including P storage in Bh horizon in nutrient management programs to reduce potential nutrient losses (Obour et al., 2011). We can see from these examples (Obour et al., 2011; Sacco et al., 2003; Watson et al., 2002) that nutrient budget calculations at various scales can help identify potential sources of nutrients by identifying areas with nutrient surplus, and reduce nutrient loss to environment by helping drive management decisions that better balance nutrient inputs and outputs.

The majority of nutrient balance studies were focused on calculation of nutrient mass balance for farms and croplands (Buciene et al., 2003; Lanyon et al., 2006; Lord et al., 2002; Slaton et al., 2004). A few studies calculated nutrient budgets in watersheds near urban areas (Watson et al., 1981), and urban areas at regional ecosystem level (Baker et al., 2001). Watson et al. (1981) determined the impact of

urbanization on Lake Wingra, WI (USA) (approximately 75% urban and 25% wooded natural) by calculating a nutrient budget (N and P). The results indicated the process of urbanization had led to a significantly increase of runoff events, which subsequently made surface runoff the dominated form of P inputs to the watershed with a decreasing ratio of N to P in annual nutrient loads. The authors stressed the importance to reduce runoff event in urban system to minimize the P movement to watershed in this area (Watson et al., 1981). Baker et al. (2001) used a nutrient budget approach and determined that food and fertilizer were the dominated forms of N inputs to urban and agricultural (cropland and dairy) system, which contributed to 88% of total N inputs to the Central Arizona-Phoenix ecosystem and were an order of magnitude higher N inputs than nearby natural settings (e.g., desert). We believe that the calculating of nutrient budgets in urban landscape can serve the same function as in agricultural system to help identify areas with nutrient surpluses and guide the development BMPs to control possible NPS pollution coming from urban system. However, the application of nutrient budgets in urban or residential landscapes at a small geographical scale (e.g., lands with turfgrass and ornamental plants) is limited in the literature.

Research Objectives

Nutrient losses from urban landscapes have become a great concern to water quality control. Compared with agricultural system, less information was available for the risk of nutrient losses in urban areas, especially the understanding of nutrient loss potential from matured mixed-species landscape. Nutrient budget calculation has proved to be an effective tool to identify areas with nutrient surplus, and subsequently guide the development of BMPs to control NPS pollution in mainly agricultural system. However, few researchers applied this tool to evaluate nutrient flows in urban green

areas within a small geographical scale. As a result, the main objective of this study was to determine the effects of plant cover (% turfgrass vs. woody ornamental) on nutrient leaching from urban landscapes by quantifying leaching N (total Kjeldahl N, NH_4–N, and nitrate–N (NO_3+NO_2–N)) and P (total Kjeldahl P and DRP). A secondary objective of this project was to calculate the soil surface nutrient budgets for mixed landscape plots containing varying proportions of turfgrass and woody ornamental species.

CHAPTER 2
EVALUATION OF NUTRIENT LEACHING FROM MIXED SPECIES RESIDENTIAL LANDSCAPES

Introduction

Environmental contamination from nitrogen (N) and phosphorus (P) non-point source (NPS) pollution has become an important concern in many areas of the world. Global population growth and the expansion of urban areas have resulted in an increase in the amount of fertilizer applied to urban landscapes to maintain aesthetically pleasing lawns and landscapes (USEPA, 1999). Without proper nutrient management, excess nutrients can be lost from urban landscapes in runoff or leachate, and subsequently reach nearby water bodies. Excess nutrient loading to water bodies can accelerate eutrophication, which results in the stimulation algae growth or cyanobacterial mats, formation of anoxic layers within the water body (Sharpley et al., 2003), death of fish and other aquatic species, and finally deterioration of overall water quality (Correll, 1998; Howarth, 1988). Water quality degradation has been reported to result from N and P levels as low as 0.05 to 0.1 mg NO_3–N L^{-1} (Burkholder et al., 1992) and 0.05 mg PO_4–P L^{-1}(Mallin and Wheeler, 2000; Paerl et al., 1995). In addition to ecological impact, excess nutrients in water bodies can also potentially endanger human safety. For example, NO_3–N concentration of 10 mg L^{-1} has been set up as safe drinking water standard, above which drinking water may result in methemoglobinemia in infants and non-Hodgkin's lymphoma (Ward et al., 1996).

Nutrient losses in runoff are a serious problem in urban areas with significant amounts of impervious cover (Schoonover and Lockaby, 2006), compacted soils (Pitt et al., 2001), and/or finer soil textures. In contrast, leaching is more prevalent in urban

pervious landscapes (e.g., lawns) where soil texture is often sandy (Petrovic, 2004) and receive water inputs that exceed plant needs (Morton et al., 1988). For example, leaching is a major pathway of nutrient loss in Florida due to generally sandy soil texture, frequent intense precipitation events, and the use of routine irrigation. Erickson et al. (2001) reported no significant N losses from surface runoff from mixed woody ornamentals and turfgrass monoculture plots (no impervious cover), even though plots were constructed at a 10% slope and experienced frequently heavy rainfall. Similarly, Loper et al. (2012) also reported no measurable runoff from mixed woody ornamentals and turfgrass monoculture plots (no impervious cover) with a 2% grade and compacted (bulk density range: 1.7–1.9 $g \cdot cm^{-3}$) soil.

Most of the previous studies evaluating nutrient leaching from urban landscapes focused on turfgrass ((Bowman et al., 2002; Easton and Petrovic, 2004; Gross et al., 1990; Petrovic, 1990). Environmental factors such as soil texture (Petrovic, 2004), soil nutrient sorption capacity (Breeuwsma and Silva, 1992), fertilizer source, rate, and application method (Easton and Petrovic, 2004; Erickson et al., 2001; Shuman, 2002), irrigation strategy (Barton and Colmer, 2006), turfgrass species (Bowman et al., 2002), and turfgrass maturity (Miltner et al., 1996) were all reported to affect nutrient leaching from turfgrass. In contrast, few studies evaluating nutrient leaching from ornamental landscapes are available in the literature. Amador et al. (2007) showed that among ten urban landscape types, the native woodland had the lowest NO_3–N leaching losses (0.17 kg ha^{-1}), followed by flowering plants (annual and perennial) and shrubs (deciduous and evergreen) landscape(< 2 kg ha^{-1}), and then areas planted with managed turfgrass, ornamental deciduous trees, and evergreen trees (2 to 10 kg ha^{-1}).

The authors suggested that priority should be given to native woodlands, annual and perennial flowers, and deciduous and evergreen shrubs rather than turfgrass when designing and managing sustainable landscapes to protect groundwater quality (Amador et al., 2007).

Researchers have recommended the use of alternative plant materials as a substitute for turfgrass in urban residential landscape to conserve water resources and minimize environmental impacts (Hipp et al., 1993; Sacamano and Jones, 1975). The University of Florida – Institute of Food and Agricultural Sciences (UF-IFAS) Florida-Friendly Landscaping™ (FFL) program promotes landscape best management practices (BMPs) to reduce the environmental impacts associated with urban landscape management. One of the core FFL landscaping principles is ‘right plant, right place’, which often promotes a reduction in areas covered with turfgrass and increased use of trees, shrubs, and other ornamental plants (Borisova et al., 2011). However, recent studies comparing the nutrient leaching from mixed woody ornamental species and turfgrass system questioned the function of ornamental plants in urban landscapes (Cisar et al., 2004; Erickson et al., 2001, 2005; Loper et al., 2012). Erickson et al. (2001) reported greater N (NO_3–N and NH_4–N) loads in leachate exported from mixed ornamental landscape plots (mainly woody ornamentals) (48.3 kg N ha^{-1}) during three fertilizer cycles than from St. Augustinegrass monoculture (4.13 kg N ha^{-1}). Erickson et al. (2005) also reported higher leaching of P and K from mixed ornamental landscape plots (37.8 and 346 kg ha^{-1}, respectively) than that from St. Augustinegrass monoculture (22.9 and 185 kg ha^{-1}, respectively). Similarly, Loper et al. (2012) reported higher amounts of nutrients (NH_4–N, NO_3 + NO_2–N, and dissolved reactive P (DRP)) were

leached from newly planted woody ornamental plant beds than from sodded turfgrass areas in the landscape.

However, none of these studies extended past the plant establishment phase to compare vegetation effects on nutrient leaching from mature landscapes. In contrast, Erickson et al. (2008) reported that significant higher inorganic-N (NO_3–N and NH_4–N) loads were leached from mixed species landscape (5.2 kg ha^{-1}) than from turfgrass monoculture (1.3 kg ha^{-1}) during the year following plant installation; but inorganic-N in leachate had no significant difference in both landscapes in subsequent years once plant material was fully established. The authors believed that the longer establishment period of woody ornamental plants compared with turfgrass may be the reason that higher nutrient lost from mixed species landscape in year one (Erickson et al., 2008). Still, there is little additional information about nutrient leaching from mixed landscapes containing established or mature woody ornamental species. The objectives of this study were to: 1) quantify nutrient (N and P) losses from established mixed urban landscapes, and 2) determine the effects of plant cover (% turfgrass vs. woody ornamental) on nutrient leaching from urban landscapes.

Materials and Methods

Lysimeter Design

Nine landscape lysimeters (3.35 m wide × 3.96 m long) were constructed at the UF-IFAS Mid Florida Research and Education Center in Apopka, FL. Lysimeters were constructed into a hill side with only the west facing wall fully exposed (Figure 2-1). Each lysimeter was 1.45 m deep along the outside edge and 1.52 m deep in the middle. The bottom of each lysimeter was sloped towards the center with a single drain pipe exiting the wall for drainage collection. The inside of each lysimeter was painted twice

with basement wall waterproof paint. The drainage system was designed around a central junction box over the center drain hole with sock-covered 10.2 cm corrugated drain pipe extending to diagonal corners. Lysimeters were covered with approximately 61 cm layer of rock, textile cloth, and coarse sand first, which was overlain with roughly 80 cm layer of native soil (a mixture of Candler-Apopka and Tavares-Millhopper soil series). In August 2009, soil in each lysimeter was leveled to about 7.62 cm below the top of each wall in preparation for installation of plant material. Backfill soil was watered-in to insure good contact between root balls and the soil at planting. Excess soil was removed from lysimeters.

Three vegetative cover treatments containing varying proportions of turfgrass and woody ornamental cover were installed in the lysimeters as follows: 1) 90% St. Augustine turfgrass, 10% woody ornamentals; 2) 75% turfgrass, 25% woody ornamentals, and 3) 60% turfgrass, 40% woody ornamental cover. All treatments contained one *Magnolia grandiflora* L. 'D.D. Blanchard' (magnolia) in the center of lysimeter. The magnolias were transplanted from 50.8-cm root control bags (High Caliper Growing Systems, OK) on 9 Sept. 2009. The magnolias were approximately 1.83 m tall and 3.81 cm in caliper as measured at 15.2 cm above ground. An area of approximately 1 m^2 was mulched around the magnolia. The 75 and 60% turfgrass treatments also contained *Viburnum odoratissimum* Ker Gawl. (sweet viburnum) shrubs. The viburnum was transplanted from 11.4-L (#3) container on 9 Sept. 2009. The 75% turfgrass treatment which contained one hedge consisting of two shrubs, mulched 0.91 m wide and 1.98 m long, was placed in the northeast corner of the lysimeters. The 60% turfgrass treatment contained two hedges. *Stenotaphrum secundatum* (Walter) Kuntze

‘Floratam’ (St Augustinegrass) sod was cut from a sand soil on 24 Sept. 2009 and was installed to cover the remaining surface area of each lysimeter on 25 Sept. 2009. Both of the turfgrass and woody ornamental species were considered fully established by the end of May 2010.

An irrigation system was installed in each lysimeter prior to planting. Pop-up spray heads (PROS-06-10A, Hunter Industries, Inc., San Marcos, CA) were positioned at each corner and in the center along the north and south sides of each lysimeter for irrigation of turfgrass areas. Shrub and tree irrigation was delivered through a 1.91 cm black polyethylene tubing trunk line outfitted with a 172 kPa pressure regulator. The tree was irrigated using two 30.5 cm tree stakes with 189 L h^{-1} nozzles and an inverted cone spreader (Jain Irrigation Inc., Fresno, CA). Shrubs were irrigated with the same spray stake assembly using 102 L h^{-1} nozzles (Jain Irrigation Inc., Fresno, CA). One stake was placed between each plant and along both outside edges of the lysimeter wall. Irrigation valves were positioned on the outside of the west-facing wall of the lysimeter. One water meter (C700-SF, Elster-Amco, Ocala, FL) with an electronic counter (32 counts L^{-1}) was installed above each valve to measure volume of irrigation applied.

Irrigation of landscapes followed a model that considered evapotranspiration (ET_0) and plant canopy’s projected surface area in our study (Beeson, 2005). Woody species were irrigated using micro-irrigation on alternate days after transplanting till late May 2010. Beginning the 1 June 2010, turfgrass and woody ornamental irrigation was controlled using a CR10X data logger program (Campbell Scientific, Inc, Logan, UT). Reference ET_o and daily rainfall amount were calculated at midnight by the onsite weather station and transferred to the CR10X data logger controlling the system. The

weather station consisted of a LI200X pyranometer (Li-Cor Inc, Lincoln, NE), a CS215 temperature and relative humidity probe (Campbell Scientific, Inc., Logan, UT), a wind sentry set (03001, R.M. Young Co., Traverse City, MI), and a tipping bucket rain gauge (TE525, Texas Electronics, Dallas, TX) connected to the CR10X data logger. Irrigation depths were calculated by multiplying ET_o by the plant coefficient (Kp) for each species (0.73, 0.70, and 0.90 for magnolia, viburnum, and turfgrass, respectively derived from previous experiments) and the ratio of the horizontal planar projected canopy area (PCA) of each species (magnolia, viburnum, and turfgrass) relative to the total surface area of each lysimeter. The Kp values were determined based on the irrigation frequency that provided an acceptable level of aesthetic quality (i.e., no brown and curling leaves, minimize drought stress) for each species (R. Beeson and M. Dukes, personal communication, 2011) and the PCA was determined every 3 weeks and updated in the data logger program. Turfgrass irrigation began at 0500 h to minimize the time turfgrass was wet to reduce disease pressure and achieve better irrigation uniformity. Shrub and tree irrigation was initiated at 0700 h.

The irrigation depths were summed each day and added to a running total irrigation depth for each lysimeter. When the cumulative total exceeded 19.0 mm of irrigation, the irrigation depth was multiplied by the area of the turfgrass to calculate the volume of water applied for the turfgrass. The PCA of the magnolia and viburnums were multiplied independently by the cumulated irrigation depth and the totals were added and the volume of irrigation was applied equally to each species. Total irrigation volume for the magnolia and viburnums defaulted to 19 L when volumes were smaller (never happen during our study period).

Fertilizers were applied initially to the plots on 11 Mar. 2011 following UF-IFAS recommendation for basic St. Augustinegrass maintenance in central Florida (Sartain, 2007). The fertilizer rates applied to woody ornamental beds were production level rates to accelerate growth from newly installed beds to mature landscape in three years. A controlled-release complete fertilizer (18-6-8, 180 d release, Florikan, Sarasota, FL) was applied to the magnolia and each viburnum hedge at rates of 220 and 244 g m^{-2}, respectively. A water soluble complete fertilizer (‘Magic Carpet’, 16-4-8, ProSource One, Lake Alfred, FL) was applied to the magnolia and each viburnum hedge at rates of 14.0 and 13.3 g m^{-2}, respectively. Turfgrass areas received a received a granular complete fertilizer (‘Natural Green’, 8-2-8, ProSource One, Lake Alfred, FL) at a rate of 60 g m^{-2}. A second application of fertilizers was applied at the same rates to both turfgrass and woody ornamental plants on 12 Sept. 2011. As a result, the total fertilizer amount applied to treatment containing higher proportion of woody ornamentals was higher than treatment containing higher proportion of turfgrass components (Table 2-1).

Soil Sampling and Analysis

Soil samples were collected at the start of the project (7 Mar. 2011), which was two days after first leachate collection. Each lysimeter was gridded into 16 units (approximately 0.83 m^2 $unit^{-1}$) and a soil sample was collected from each unit to a depth of 15 cm. Soil samples from turfgrass areas and woody ornamental areas were collected and composited separately to create two soil samples per lysimeter. Composite samples were air-dried at 25 ± 2 ºC and sieved through 2-mm screen before analysis. Particle size distribution was determined using the hydrometer method (Bouyoucos, 1962). Soil pH (1:2 soil to deionized water ratio) and organic matter (loss on ignition) were determined by standard methods of the UF-IFAS Extension Soil

Testing Laboratory (Mylavarapu, 2009). Soil test P, K, Ca, Mg, Fe, and Al were analyzed using inductively coupled plasma – atomic emission spectroscopy (ICP-AES) (USEPA Method 6010C) after Mehlich-3 extraction (Sims, 2009). Total P in soil was analyzed using ICP-AES (USEPA Method 6010C) after acid digestion (USEPA, 1986).

Daily Leachate Volume Quantification

A dry well was installed below the 5.08-cm lysimeter drain (Figure 2-2), which consisted of an upper collection vessel with a volume of 1.5 L that drained into a weighing vessel through a normally open valve. The weighing vessel was suspended from a 22.7 kg load cell (Interface Inc., Scottsdale, AZ) with a normally closed valve in the bottom. The weighing vessel also had an overflow drain installed near the top to channel water below the drain valve should leachate exceed the capacity of the system. When a computer determined the vessel should be drained, a 12 volt (direct current) sump pump was activated to close the upper valve (normally open) and stop the flow into the weighing vessel; concurrently, the drain valve (normally closed bottom valve) was opened to evacuate the weighing vessel. Once empty, the pump power was turned off and the leachate that had drained from the lysimeter during the evacuating process was allowed to proceed to the weighing vessel. The measuring device was located inside an enclosure with a tin roof and cement board sides (Hardie board, James Hardie, Mission Viejo, CA) that were sealed with silicon and expanding foam to exclude rainfall, dust, and blowing sand.

The system determined the mass of each weighing vessel every 2 min and compared the mass to the empty vessel. When a minimum of 1 L of leachate was collected, the measuring device valves were activated and the measuring vessel was then weighed every 10 s. When all water drained out, the valves were de-activated and

the amount drained was added to a running daily total. The volume of drainage water from each lysimeter was recorded daily at 0500 h, when the running total was stored by a computer. The leachate volume was then reset to zero and daily volume determination began again.

Leachate Sampling and Analysis

Drainage water from the nine lysimeters containing established mixed landscapes was collected from 5 Mar. 2011 to 9 Mar. 2012. Water drained through the lower measuring valve flowed horizontally through 10-cm of braded polyurethane tubing (12.5 mm inner diameter) and then flowed vertically out of the tube through a 90 degree polyethylene elbow. Leachate was sampled by inserting a 16G 1 needle (Becton Dickinson Co., Rutherford, NJ) through the bottom of the polyurethane tubing. As water flowed through the tubing, 1.0 ± 0.2% of the total drainage water passed through the needle and collected in an open top 3.0-L graduated beaker. Leachate collected in these beakers was retrieved daily at 0745 h (Monday through Friday) to serve as the daily subsample. Weekend subsamples were collected the following Monday. Daily leachate subsamples were stored frozen until they were combined to produce a weekly flow-weighted composite sample (based on total volume).

Leachate samples were filtered through a 0.45-µm filter and then split into four equal portions for analysis of N and P. One sample was digested using modified USEPA Method 351.2 (USEPA, 1993b) and analyzed for total Kjeldahl N (TKN) and total Kjeldahl P (TKP) using a discrete analyzer (AQ2, Seal Analytical, West Sussex, UK). The modified method employed a 5 to 1 sample to solution ratio instead of the 2.5 to 1 sample to solution ratio outlined in USEPA Method 351.2. Two leachate sample portions were analyzed for NH_4–N (USEPA, 1993a) and NO_3+NO_2–N (NO_x) (USEPA,

1993c) colorimetrically using a discrete analyzer (AQ2, Seal Analytical, West Sussex, UK). The last portion of leachate was analyzed using molybdate-blue reaction to determine DRP (Murphy and Riley, 1962). Nutrient loads were calculated by multiplying the total leachate volume by the nutrient concentration. Cumulative nutrient loads were calculated by adding up nutrient loads over the total one year collecting period (5 Mar. 2011 to 9 Mar. 2012) for each of the mixed landscape treatments.

Statistical Analysis

The experiment was designed as a randomized complete block design with three vegetation treatments applied in three blocks (replicates) (each block consisting of a set of three consecutive lysimeters). Missing leachate volume and nutrient data were estimated by random generation based on the sample mean using the rannor function in SAS using PROC SQL (Zhang et al., 2008). All data except DRP and NH_4–N concentration were log transformed before statistical analysis. Leachate volume, flow-weighted weekly nutrient concentrations, weekly loads, and cumulative data were analyzed using the PROC MIXED procedure in SAS with landscape treatment as a fixed effect and block as a random effect (SAS Institute, 2008) using a repeated measures model with ar(1) covariance structure (except cumulative data). Normality was checked by examining histogram and normality plots of the conditional residuals. Leachate volume, nutrient concentrations, and loads were analyzed separately for each week when a significant treatment × time interaction was noted. Both DRP and NH_4–N concentrations were analyzed by week to better fit the normality assumption. All pairwise comparisons were completed using the Tukey's honestly significant difference test with a significance level of $\alpha = 0.05$.

Results and Discussion

Initial Soil Properties

Particle size analysis revealed that soils from both turfgrass and woody ornamental plants had sandy texture (data not shown). Soils collected from the turfgrass areas had significantly higher pH (6.62) than soils collected from woody ornamental plant beds (6.21). The pH of turfgrass and woody ornamental soils were close to the targeted pH for selected plant species (turfgrass: 6.5; woody ornamental plants: 6.0) that well suited to Florida growing conditions (Kidder et al., 1998). Soils under turfgrass had higher organic matter content (0.90%) than soils in woody ornamental plant beds (0.64%), which can be explained by the relatively high accumulation rate of organic matter under turfgrass areas due to quick root turnovers (Qian and Follett, 2002). Soil nutrient analysis indicated that soils from turfgrass areas had higher concentrations of Mehlich 3 P, K, Ca, Mg, and Fe than soils under woody ornamental plants (Table 2-2). Mehlich 3 P and K levels in our soils under turfgrass were medium (conversion from Mehlich 1 to Mehlich 3 using the relationship reported in (Mylavarapu et al., 2002), in which case the probability of plant response to applied fertilization would be ≤25% (Sartain, 2008). In contrast, soil interpretations for P and K levels about soils under woody ornamental plants were low and very low, respectively (Kidder et al., 1998), suggesting that the probability of plant response to applied fertilization would be ≤50% and ≤75% for P and K, respectively (Sartain, 2008). The difference in initial nutrients content between soils under turfgrass and woody ornamental plants may due to the improved soil water holding capacity under turfgrass related to higher storage of soil organic matter (Khaleel et al., 1981), which subsequently reduces the mobility of nutrients dissolved in water, or higher nutrient uptake capacity of established

ornamental plants results from their higher root (Jackson et al., 1996) and aboveground biomass compared with turfgrass.

Rainfall, Irrigation, and Leachate Volume

The average cumulative irrigation applied to lysimeters containing the 60, 75, and 90% turfgrass treatments was 6,796, 8,821, and 12,142 L per lysimeter (area = 13.26 m^2), respectively, indicating that more irrigation was applied to the landscape treatments with a higher proportion of turfgrass (Figure 2-3). This was mainly due to the fact that higher Kp value of turfgrass (0.9) than woody ornamental plants (0.73 and 0.70 for magnolia and viburnum respectively) was used in our study to determine the irrigation depth. The higher irrigation inputs to treatments containing higher proportion of turfgrass was also in agreement with the fact that turfgrass generally require higher irrigation than ornamental plants to maintain a similar visual quality level (Pittenger et al., 2001).

The average ratio of drainage volume to the sum of rainfall and irrigation over the course of the study was 0.13, 0.15, and 0.37 for the 60, 75, and 90% turfgrass landscape treatments, respectively. The 90% turfgrass treatment leached almost two times more of the applied irrigation and rainfall than the 60 and 75% turfgrass landscape treatments. Our results suggested that landscapes containing higher proportion of woody ornamentals to turfgrass had better water use efficiency. The ratio of drainage volume to rainfall + irrigation applied exceeded 1.0 for all three treatments at weeks 5 and 19, and 90% turfgrass treatment only at weeks 21, 36, and 52 (Figure 2-3), which reflects a lag time effect for the drainage of applied irrigation and rainfall through the 1.52 m deep lysimeters (from surface to bottom). In other words, the applied water to the surface of soils would be delayed to show up as leachate in the bottom of lysimeter soils. Similarly, studies have shown that dissolved pollutants (e.g., NO_3) in

surface soils would not immediately reach, and therefore, contaminate the underlying aquifer (Jury et al., 1991). The time needed for water to move from surface soils to groundwater depended on factors like climate, soil properties, vertical distance from soil surface to the aquifer, and agricultural practices (Jury et al., 1991). Researchers have empirically regarded this phenomenon as “effect of soil transport time lags” or “leaching time lags” (Fleming et al., 1995). Total rainfall and irrigation input were always high immediately prior to drainage events where the rainfall + irrigation to drainage ratio exceeded 1.0.

The volume of leachate collected from the mixed landscape lysimeters was mainly influenced by the amount of rainfall received. Higher leachate volumes were collected during weeks when higher total amount of precipitation occurred (Figure 2-3). A significant time × treatment effect was noted for the leachate volume. When evaluating the vegetation effect by week, significant differences among treatments on leachate volume were recorded for weeks 13, 24, 25, 30, 35, 48, and 51 through 53; there was no significant vegetation effect on leachate volume at any other week during our study (data not shown). In general, the 90% turfgrass treatment generated significantly higher leachate volumes than treatments with 60 and/or 75% turfgrass treatments (Table 2-3). No plant cover treatment effect was noted for cumulative leachate volume collected from mixed landscape lysimeters, but numeric trends showed that leachate volume increased with increasing proportion of turfgrass (Table 2-4). This may because of the fact that our experiment was a field experiment with just three replicates. Also, besides precipitation, cumulative leachate volume was also affected by irrigation. Ideally, the computer-controlled irrigation condition for each lysimeter within the same treatment

should be exactly the same. But in reality, the irrigation received by each replicate may be affected by micro-climate, plants' growing condition and other factors of each lysimeter, which may result in relatively high variability among replicates and mask the difference among treatments on leachate volume. However, 90% turfgrass treatment still had significantly higher cumulative leachate volume than 60% turfgrass treatment (P value = 0.05) with a significant level $\alpha = 0.1$.

Our results contradict those of Erickson et al. (2001), who showed that mixed species landscapes (mainly woody ornamentals) drained more water annually (2,237 mm) than St. Augustinegrass monoculture (2,082 mm) during the first year after planting. Loper et al. (2012) also reported higher leachate volumes collected from landscapes containing woody ornamental plants (mean = 2,710 L ha^{-1}) than from St. Augustinegrass monoculture (mean = 951 L ha^{-1}) during a one-year plant establishment study. Our results were in agreement with those of Erickson et al. (2005), who reported that the seasonal drainage volume was significantly smaller from established mixed species (2-years old) (24.2 mm in dry season and 393 mm in wet season) than from turfgrass monoculture (59.4 mm in dry season and 515 mm in wet season). The lower irrigation inputs and lower leachate volume from landscape treatment containing a higher proportion of woody ornamental plants in our study may be explained by the ability of woody ornamental plants to absorb water from deeper within the soil profile compared with the more shallow-rooted turfgrass (Jackson et al., 1996; Nepstad et al., 1994). For example, Jackson et al. (1996) reported that 44% of grass roots were distributed in the top 10 cm of soil. In contrast, temperate and tropical tree species had 26 and 60% of their root scattered in the top 10 and 30 cm of soil respectively. The

deep-rooted system of woody ornamental plants can make full use of stored soil moisture in deeper soil depth (Pittenger et al., 2001).

Leachate Nutrient Concentrations and Loads

Mixed landscape treatments affected the concentration of TKN and NO_x in leachate throughout the study (Table 2-5). Leachate collected from the 90% turfgrass treatment contained significantly higher mean TKN and NO_x concentrations than the 60 and 75% turfgrass treatments (Table 2-5). The overall mean concentration of NO_x ranged from 0.1 to 1.05 mg L^{-1} among different landscape treatments and never exceeded the 10 mg L^{-1} NO_3–N drinking water standard. But leachate NO_x concentrations sometimes exceeded levels (i.e., 0.05 – 0.1 mg NO_3–N L^{-1}) that can potentially result in water quality degradation (Burkholder et al., 1992).

A significant vegetation treatment × date effect was noted for leachate NH_4–N concentrations (data not shown); therefore, leachate NH_4–N concentrations were evaluated on a weekly basis. A significant vegetation effect on leachate NH_4–N concentration was noted at 2, 17, 47, 49, 52, and 53 weeks (Figure 2-4). In those weeks, NH_4–N concentrations in leachate from the 90% turfgrass treatment were significantly higher NH_4–N than in leachate collected from the 60 and/or 75% turfgrass treatment (Figure 2-4). There was no plant cover treatment effect on leachate NH_4–N concentrations at any other time during our study (data not shown); mean leachate NH_4–N concentration during weeks with no treatment effect was 0.05 mg L^{-1}.

Analysis of leachate samples for the first 18 weeks showed that the concentrations of DRP and TKP in the same samples were not significantly different (data not shown). Therefore, we suggest that very little unreactive P (TKP – reactive P) (mainly in the form of organic P) was leached from the mixed landscape lysimeters, which was probably

related to the fertilizer source (inorganic only) applied to the lysimeters. Toor et al. (2004) reported that the form of P applied influenced the forms of leached P; more total P (TP) and unreactive P leached from intact soil monoliths (taken from a permanent grassland site) receiving a combination of mineral P fertilizer and farm dairy effluent than soil monoliths receiving mineral fertilizer alone when applied at similar TP rates (mineral alone: 90 kg ha^{-1}, mineral with farm dairy effluent: 75–86 kg ha^{-1}). The authors suggested that the increase in TP and unreactive P in leachate was likely due to the fact that unreactive organic P species in the farm dairy effluent (manure and feces typically contained around 25% TP as organic P (Toor et al., 2006)) was prone to movement through the soil profile (Toor et al., 2004). Analysis of leachate TKP concentration ceased at 18 weeks due to the absence of significant concentrations of unreactive P.

Mean concentrations of DRP ranged from 0.00 to 0.41 mg L^{-1} over the course of our study. Many samples had DRP concentrations that were close to or below the detection limit for P (0.088 mg L^{-1}). A significant vegetation treatment × date effect was noted for DRP concentration in leachate; therefore, analysis was conducted by sampling date. Leachate DRP concentrations were affected by mixed landscape treatments in weeks 2, 8, 10, 21, 23, 24, 25, 30, 45, 47, and 53 of the study (Table 2-6), where leachate DRP concentrations collected from the 90% turfgrass treatment were significantly higher than leachate DRP concentrations collected from lysimeters containing the 60 and/or 75% turfgrass treatments (Table 2-6). An exception to this trend was noted only in week 2, when lysimeters planted with the 60% turfgrass treatment had significantly higher leachate DRP concentrations than lysimeters planted with the 90% turfgrass treatment (Table 2-6), which might result from the unstable

behavior of testing instrument when DRP concentration was close to or below the detection limit. There was no plant cover treatment effect on leachate DRP concentrations during any other week of our study (data not shown); mean leachate DRP concentration was 0.04 mg L^{-1} for sampling dates with no mixed landscape treatment effect.

A significant date × vegetation treatment interaction was noted for TKN, NO_x–N, NH_4–N, and DRP loads leached from mixed landscape plots; therefore, vegetation effects were analyzed separately for each week. When statistical analysis indicated a significant vegetation effect, the 90% turfgrass treatment typically leached significantly higher TKN, NO_x–N, NH_4–N, and DRP loads than the 60 and/or 75% turfgrass treatments (Table 2-7; Figures 2-5 to 2-7). No treatment effect was noted for TKN, NO_x–N, NH_4–N and DRP loads in leachate at any other time during our study (data not shown). Mean TKN, NO_x–N, NH_4–N and DRP loads for weeks without treatment effect were 85.0, 42.6, 4.28, and 4.45 g ha^{-1}, respectively.

Cumulative TKN, NO_x–N, and NH_4–N loads leached from mixed landscape lysimeters were affected by vegetation treatments (Table 2-4). In general, cumulative N loads (all measured forms) leached from the 90% turfgrass treatment were significantly higher than cumulative N loads leached from the 60 and 75% turfgrass treatments (Table 2-4). Leachate loads of NO_x (0.71 to 4.77 kg ha^{-1}) were significantly higher than NH_4–N loads (0.15 to 0.48 kg ha^{-1}) from all three mixed landscape treatments in our study. This was consistent with the fact that NH_4–N is generally considered to be less mobile than NO_x–N due to the cation exchange reactions with the soil (Bowman et al., 2002).

The total cumulative DRP loads exported from our study were very low (0.1-0.39 kg ha^{-1}). Soldat and Petrovic (2008) reported that annual P leaching losses from sandy soils growing with fertilized turfgrass ranged from 0.03 to 6.1 kg ha^{-1}. This may because of the relatively high Mehlich 3 extractable Al and Fe concentrations in soils (Table 2-2), which limit the P mobility by chemical bonding with P (Borggaard et al., 2005; Tunesi et al., 1999). For example, Kleinman and Sharpley (2002) used 37 acidic and 25 alkaline surface soils obtained from the USDA Natural Resources Conservation Service's National Soil Survey Laboratory archives to estimate soil P sorption saturation. The highest concentration of Al and Fe of those soils was 1.43 and 0.41 mg kg^{-1} respectively, which was lower than the concentration of our soils (Table 2-2). However, Nair et al. (2004) reported that the change point of degree of P saturation (DPS) ((M3-P)/α(M3-Fe+M3-Al))×100, where M3-P, M3-Fe and M3-Al is the concentration of Mehlich-3 extractable P, Al, and Fe expressed in mg kg^{-1}, α is an empirical factor considering P saturations of different soils) for Florida sandy soil was 16%, above which water soluble P in soils and P loss potential increased significantly. The DPS values of our soils under turfgrass and woody ornamental plants were 38 and 27% respectively (both were higher than the reported threshold) when following the calculation by Nair et al. (2004), suggesting that our soils have relatively low ability to hold additional P and P losses potential would be high after fertilization. As a result, we suspected that the majority of applied P in our study was absorbed and removed by the fully established plants. For example, Soldat and Petrovic (2008) summarized that P removal in turfgrass clipping could be as high as 2 to 15 kg ha^{-1} yr^{-1} in temperate climates. The intense precipitation and warm temperature in Florida can stimulate the growth of plants in our study and

would potentially absorb a significant amount of P from soil water, which subsequently reduced leaching loss potential.

No plant cover treatment effect was noted for cumulative DRP load exported, but cumulative DRP load increased numerically with increasing proportion of turfgrass (Table 2-4). However, similar to cumulative leachate volume, 90% turfgrass treatment had significantly higher cumulative DRP load exported than 60% turfgrass treatment (P value= 0.08) with a significant level α=0.1.

Our results differed from those of Erickson et al. (2001), who reported a that NO_3–N loads in leachate were significantly greater from a mixed-species woody ornamental landscape (48.3 kg ha^{-1} NO_3–N) than from St. Augustinegrass monoculture (4.13 kg ha^{-1} NO_3–N) during the first year after planting. Similarly, Erickson et al. (2005) reported higher cumulative mean P loads in leachate collected from mixed woody ornamental species (37.8 kg ha^{-1}) than turfgrass monoculture (22.9 kg ha^{-1}) during the same time period. Cisar et al. (2004) also reported that N leaching concentration and loads (both NO_3–N and NH_4–N) from woody ornamental plants (e.g., annual mean NO_3–N and NH_4–N concentration 1.46 and 0.3 mg L^{-1}) were higher than turfgrass (e.g., mean NO_3–N and NH_4–N concentration <0.2 and <0.3 mg L^{-1}), even though woody ornamental plants required half the fertilizer inputs of St. Augustinegrass. In another study, Loper et al. (2012) reported higher nutrient (NH_4–N, NO_3 + NO_2–N, and DRP) leaching (both concentration and load) from newly planted woody ornamental plant beds than from turfgrass areas in the landscape during a one year establishment study.

The discrepancy between our results and leachate volume and nutrient loss (concentration and load) results published in the literature (Cisar et al., 2004; Erickson

et al., 2001; Erickson et al., 2005; Loper et al., 2012) from woody ornamental species and turfgrass was likely due to the fact that the referenced studies measured leachate volume and nutrient export during the plant establishment period (less than one year after planting). Erickson et al. (2008) showed that establishing mixed woody ornamental species had lower root biomass (100 g m^{-2}) and density than turfgrass (450 g m^{-2}), which made the plants less efficient at removing water and nutrients from the soil. As a result, the lower water and nutrient use efficiency during establishment is related to the root systems of woody ornamental plants following transplant. Erickson et al. (2008) attributed the higher nutrient loss in leachate from mixed species landscape in year one to relatively longer establishment time that was required for woody ornamental plants. For example, Wiese et al. (2009) reported the root of *Ilex cornuta* 'Burfordii Nana' and *Pittosporum tobira* 'Variegata' shrubs did not reach beyond the edge of the canopy until about 20 weeks after planting (WAP). As a result, roots might be absent in the spaces between newly planted shrubs, which increases the chance of applied fertilizers lost from these areas. Wiese et al. (2009) also reported that the root spread radius (1/2 × (east root length+ west root length)) and biomass of both species almost doubled from 52 to 90 WAP under three irrigation frequencies. In another study, Gilman (1990) recorded the root spread diameter of caliper live oak increased from 13 cm before transplanting to 6.1 m at 2 years after transplanting in Florida. In general, it can take a longer time for woody ornamentals to colonize large areas in landscape than turfgrass. In our study, both of the woody ornamental species were planted in the lysimeters for a period of 1.5 years before leachate sample collection commenced. Therefore, the root systems of the woody ornamental plants would have more root spreading and biomass

(Gilman, 1990; Wiese et al., 2009) compared with woody ornamental plants during the plant establishment period (as reported by Cisar et al. 2004; Erickson et al. 2001; Erickson et al. 2005; Loper et al. 2012) Increased root colonization of the soils in our landscape lysimeters explains the higher water use efficiency and lower nutrient (both N and P) losses from landscape treatments containing a higher proportion of woody ornamental species in our study.

Our findings were supported by Amador et al. (2007), who compared N losses from 10 types of urban landscapes including different types of ornamental plants (e.g., ground covers, annuals, woody shrubs, etc.) and turfgrass over a 18-months study. The authors reported that the risk of NO_3–N leaching in areas planted with managed turfgrass (7.46 kg ha^{-1} NO_3–N) was similar to areas planted with ornamental deciduous and evergreen trees (7.14 and 9.03 kg ha^{-1} NO_3–N, respectively), but was higher than areas planted with deciduous and evergreen shrubs (1.53 and 1.70 kg ha^{-1} NO_3–N, respectively). In another study, Erickson et al. (2008) reported significantly higher amount inorganic–N (NH_4 + NO_3–N) leaching load was exported from mixed species landscape than turfgrass monoculture in year one when woody ornamental species were still expanding. But no significant difference in the amount of leaching inorganic-N was observed between two types of landscape once the woody ornamental species were more established in year two, suggesting an improved nutrient absorbing ability of established woody ornamental plants.

The lower leachate volumes and nutrient losses from landscapes containing higher proportions of woody ornamental plants may also be explained by the efficient cooperation of shallow root system (turfgrass) with deep root system (woody

ornamental plants) to absorb water and nutrients from different soil depths (Nair and Graetz, 2004; Nair et al., 2007). Nair et al. (2007) noted that silvopasture systems (combination of pasture and tree) had lower water soluble P concentration in soil (mean: 2.51 mg kg^{-1} in the surface to 0.087 mg kg^{-1} at 1.0 m depth) than treeless pasture systems (mean: 9.11 mg kg^{-1} in the surface to 0.23 mg kg^{-1} at 1.0 m depth) throughout the soil depth, measured to a depth of 1 m. The authors attributed the results to the enhanced nutrient uptake by extensive root system of trees from different soil depths (Nair et al., 2007).

In summary, our study indicated that increasing the composition of established woody ornamental plants in mixed urban landscape decreased the volume of leachate, which in turn led to reduced nutrient losses. Therefore, we recommend that priority should be given to landscape types with higher proportion of woody ornamental plants when designing and managing urban green areas; this will help to improve water use efficiency and reduce potential NPS pollution from home landscapes. Our recommendations are consistent with current landscape BMPs promoted by the UF-IFAS Florida Friendly Landscaping™ Program, specifically the “right plant, right place” principle. Although, woody ornamental plants can have higher nutrient loss potential than turfgrass during establishment (Cisar et al., 2004; Erickson et al., 2001, 2005; Loper et al., 2012), our results suggested that incorporation of woody ornamental plants may create a more environmental-friendly landscape in the long-term once the root systems of woody ornamentals have developed after establishment. However, our study evaluated only one warm-season turfgrass (St. Augustinegrass) in the mixed urban landscapes. Future research could compare the nutrient leaching from mixed urban

landscape considering other warm-season turfgrass species (e.g., zoysiagrass and bahiagrass) or cool-season turfgrass (for other areas in U.S.) and ornamental species (e.g., other woody species, herbaceous species) to find out the most environmental-friendly combination of turfgrass and ornamental plants for sandy soils in urban areas.

Table 2-1. Annual fertilization rate of N, P, and K to drainage lysimeters plots containing three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10).

Treatment	N	P	K
	g m^{-2}		
60% Turfgrass	18.6	2.60	8.29
75% Turfgrass	13.1	1.77	6.59
90% Turfgrass	7.63	0.95	4.89

Table 2-2. Mean and standard deviation for chemical properties of soils collected at the onset of leachate collection (7 Mar. 2011) from turfgrass and woody ornamental areas of drainage lysimeters containing three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10).

Parameter	Turfgrass	Ornamental
	mg kg^{-1}	
Total P	123 ± 13.2 b[a]	98.1 ± 36.7 a
Mehlich 3 P	59.2 ± 9.70 b	40.4 ± 21.9 a
Mehlich 3 K	42.4 ± 7.40 b	18.7 ± 14.2 a
Mehlich 3 Ca	375 ± 48.5 b	200 ± 67.3 a
Mehlich 3 Mg	68.3 ± 11.2 b	40.8 ± 10.2 a
Mehlich 3 Fe	61.8 ± 9.72 b	43.7 ± 12.3 a
Mehlich 3 Al	250 ± 33.7 a	261 ± 51.7 a

[a]Values within the same test parameter with the same letter are not significantly different at $P < 0.05$ using Tukey's HSD test.

Table 2-3. Mean weekly leachate volume from lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10) during sampling dates where there was a significant plant cover treatment effect.

Week	60% Turfgrass	75% Turfgrass	90% Turfgrass
	kL ha^{-1}		
13	0.00 a[a]	1.24 ab	154 b
24	0.00 a	24.2 ab	119 b
25	0.00 a	12.2 ab	54.7 b
30	0.00 a	1.74 ab	62.7 b
35	0.00 a	0.03 ab	0.09 b
48	0.00 a	0.00 a	33.9 b
51	0.00 a	0.00 a	52.9 b
52	0.00 a	0.00 a	100 b
53	0.00 a	0.00 a	43.4 b

[a]Values within the same week with the same letter are not significantly different at $P < 0.05$ using Tukey's HSD test.

Table 2-4. Cumulative volume and mass of total Kjeldahl N (TKN), NO_3+NO_2–N (NOx), NH_4–N, and dissolved reactive P (DRP) leached from lysimeters planted with three mixed landscape treatments containing turfgrass and woody ornamental at proportions (%) of 60/40, 75/25 and 90/10 from 5 Mar. 2011 to 9 Mar. 2012).

Treatment	Volume	TKN	NO_x	NH_4	DRP
	million L ha^{-1}	kg ha^{-1}			
60% Turfgrass	1.91 a[a]	2.75 a	1.81 a	0.15 a	0.14 a
75% Turfgrass	2.64 a	3.75 a	0.71 a	0.19 a	0.18 a
90% Turfgrass	5.14 a	8.49 b	4.77 b	0.48 b	0.39 a

[a]Values within the same test parameter with the same letter are not significantly different at P < 0.05 using Tukey's HSD test.

Table 2-5. Mean flow-weighted leachate total Kjeldahl N (TKN) and NO_3+NO_2–N (NOx) concentrations collected from lysimeters planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10).

Treatment	TKN	NO_x
	mg L^{-1}	
60% Turfgrass	0.35 a[a]	0.26 a
75% Turfgrass	0.68 a	0.10 a
90% Turfgrass	1.22 b	1.05 b

[a]Values within the same test parameter with the same letter are not significantly different at P < 0.05 using Tukey's HSD test.

Table 2-6. Mean weekly dissolved reactive P (DRP) concentrations in leachate from lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10) during weeks when there was a significant plant cover treatment effect.

Week	60% Turfgrass	75% Turfgrass	90% Turfgrass
	mg L^{-1} DRP		
2	0.09 b[a]	0.05 ab	0.03 a
8	0.00 a	0.01 ab	0.08 b
10	0.00 a	0.02 ab	0.07 b
21	0.01 a	0.07 b	0.08 b
23	0.00 a	0.03 ab	0.09 b
24	0.00 a	0.04 ab	0.09 b
25	0.00 a	0.03 a	0.10 b
30	0.00 a	0.02 a	0.09 b
45	0.00 a	0.00 a	0.07 b
47	0.00 a	0.00 a	0.06 b
53	0.00 a	0.00 a	0.06 b

[a]Values within the same week with the same letter are not significantly different at P < 0.05 using Tukey's HSD test.

Table 2-7. Mean weekly NO_3+NO_2–N (NOx) loads from lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10) during weeks with significant plant cover treatment effect.

Week	60% Turfgrass	75% Turfgrass	90% Turfgrass
	g NO_x ha^{-1}		
13	0.00 a[a]	0.27 a	83.2 b
17	0.24 a	2.72 a	249 b
21	25.4 a	3.54 a	386 b
23	0.00 a	0.28 ab	38.6 b
24	0.00 a	1.78 a	198 b
25	0.00 a	0.59 ab	69.4 b
30	0.00 a	0.20 a	75.0 b
32	45.8 ab	6.82 a	507 b
35	0.00 a	0.47 ab	58.2 b
52	0.00 a	0.00 a	119 b
53	0.00 a	0.00 a	61.2 b

[a]Values within the same week with the same letter are not significantly different at $P < 0.05$ using Tukey's HSD test.

Figure 2-1. Overview of lysimeter design and layout as seen when looking north between the two rows of drainage lysimeters at the UF-IFAS Mid Florida Research and Education Center in Apopka, FL. Picture taken July 2010.

Figure 2-2. Measuring device for quantifying leachate from each lysimeter at the UF-IFAS Mid Florida Research and Education Center in Apopka, FL. Picture taken July 2010.

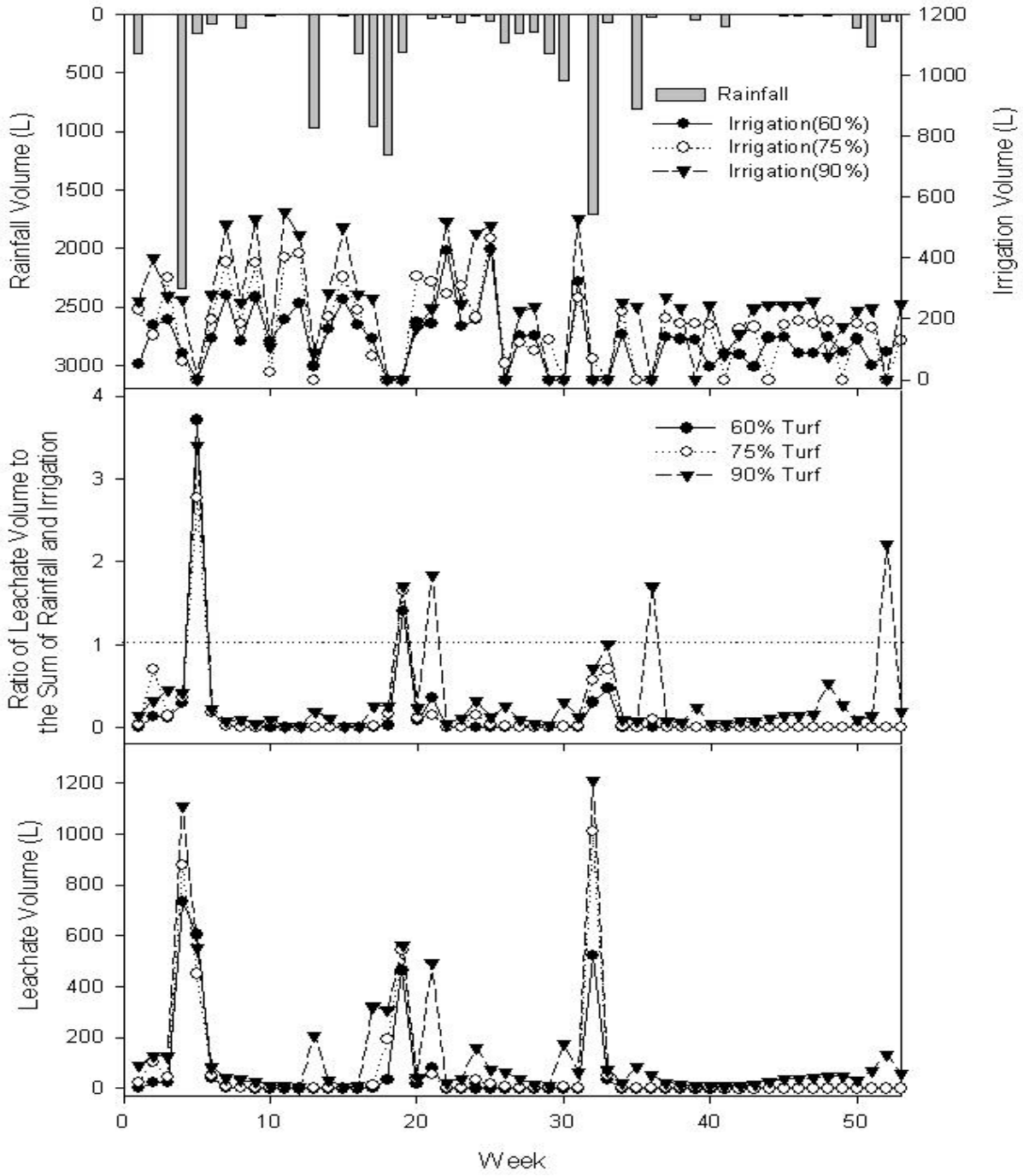

Figure 2-3. Temporal trends in mean weekly applied rainfall and irrigation volume (upper), the ratio of leachate volume to the sum of rainfall and irrigation (middle), and the mean weekly leached drainage volume from lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10).

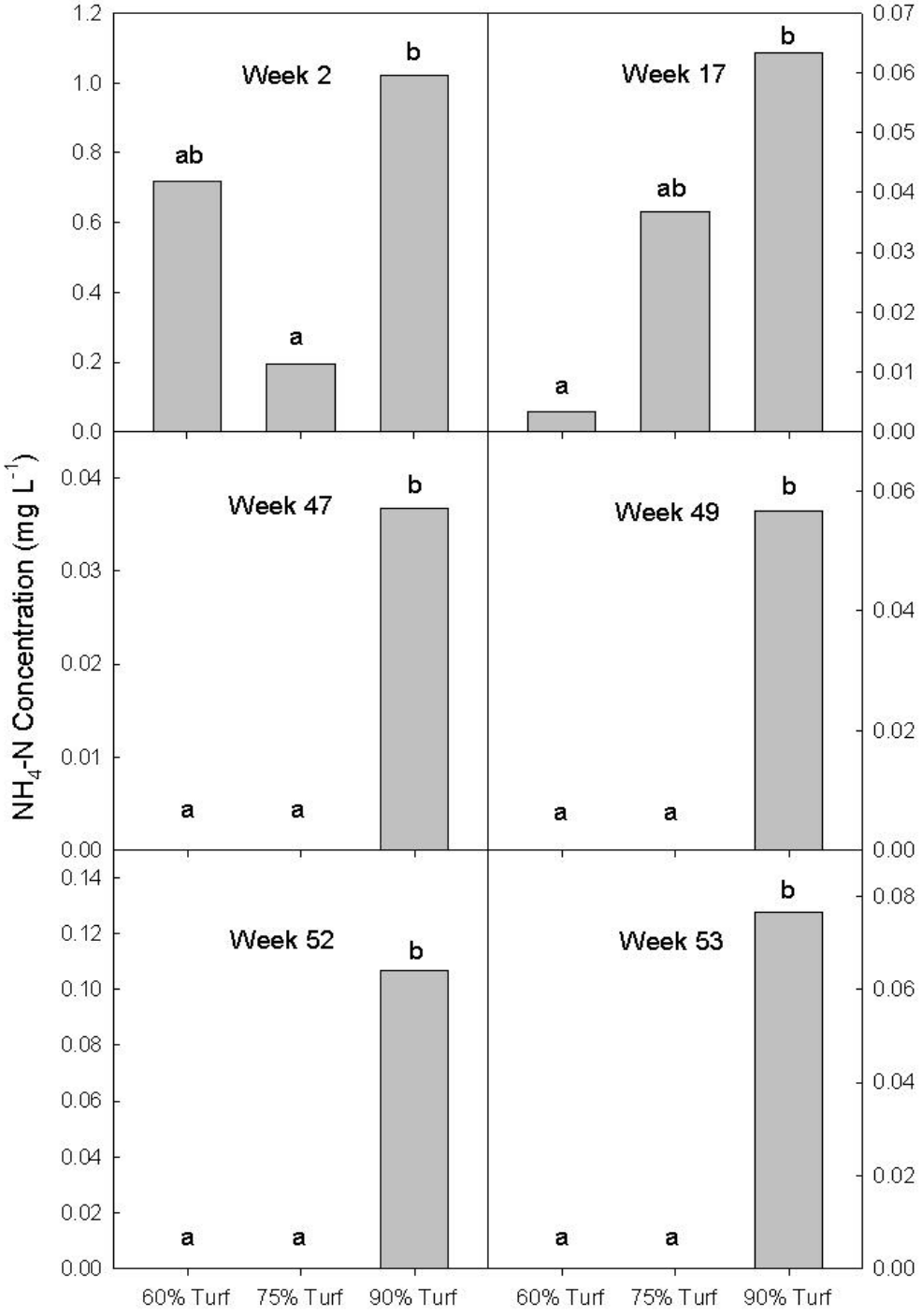

Figure 2-4. Mean weekly NH_4–N concentrations in leachate from lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10) from weeks with significant plant cover treatment effect.

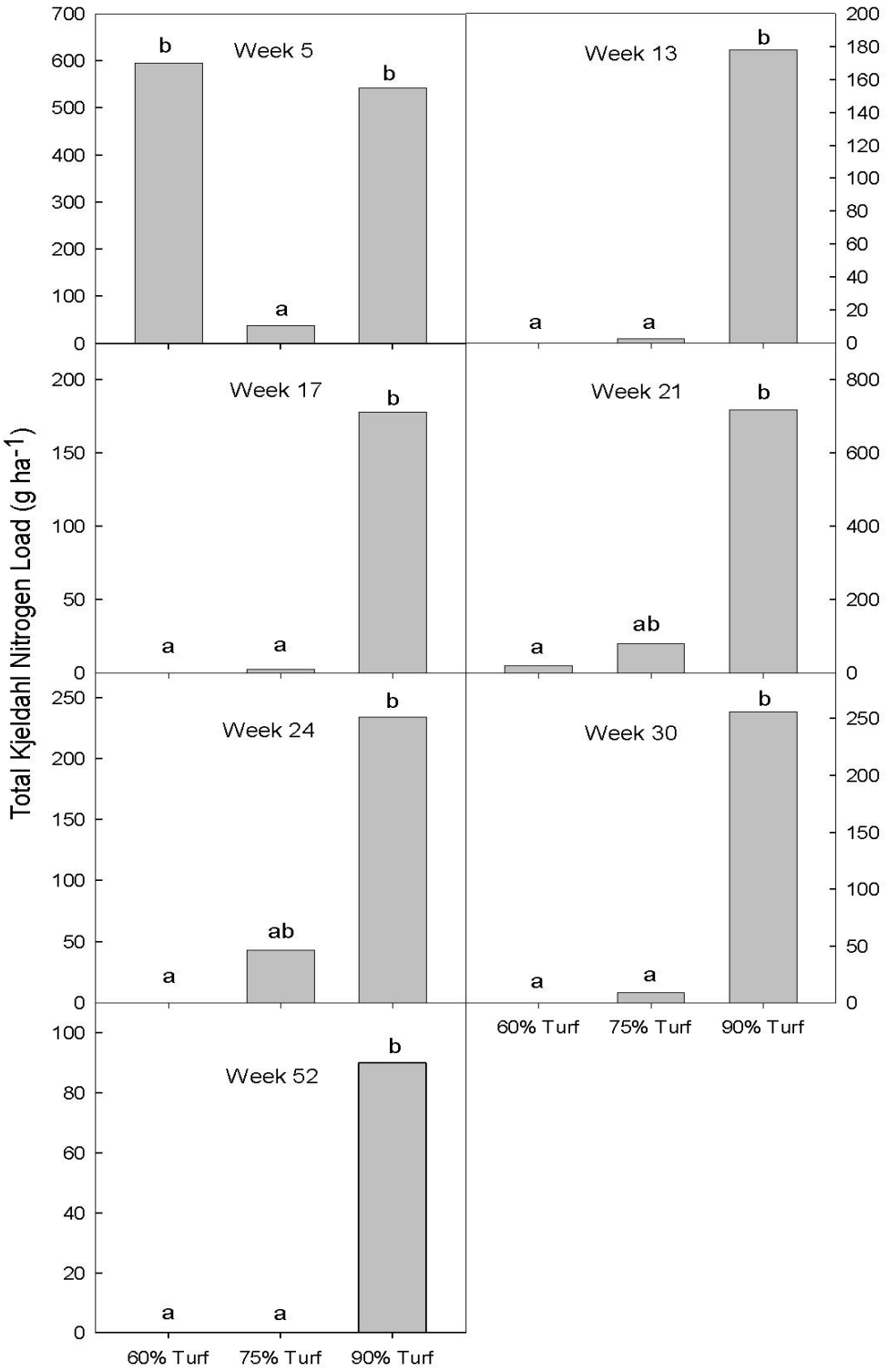

Figure 2-5. Mean weekly total Kjeldahl N loads from lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10) during weeks with a significant plant cover treatment effect.

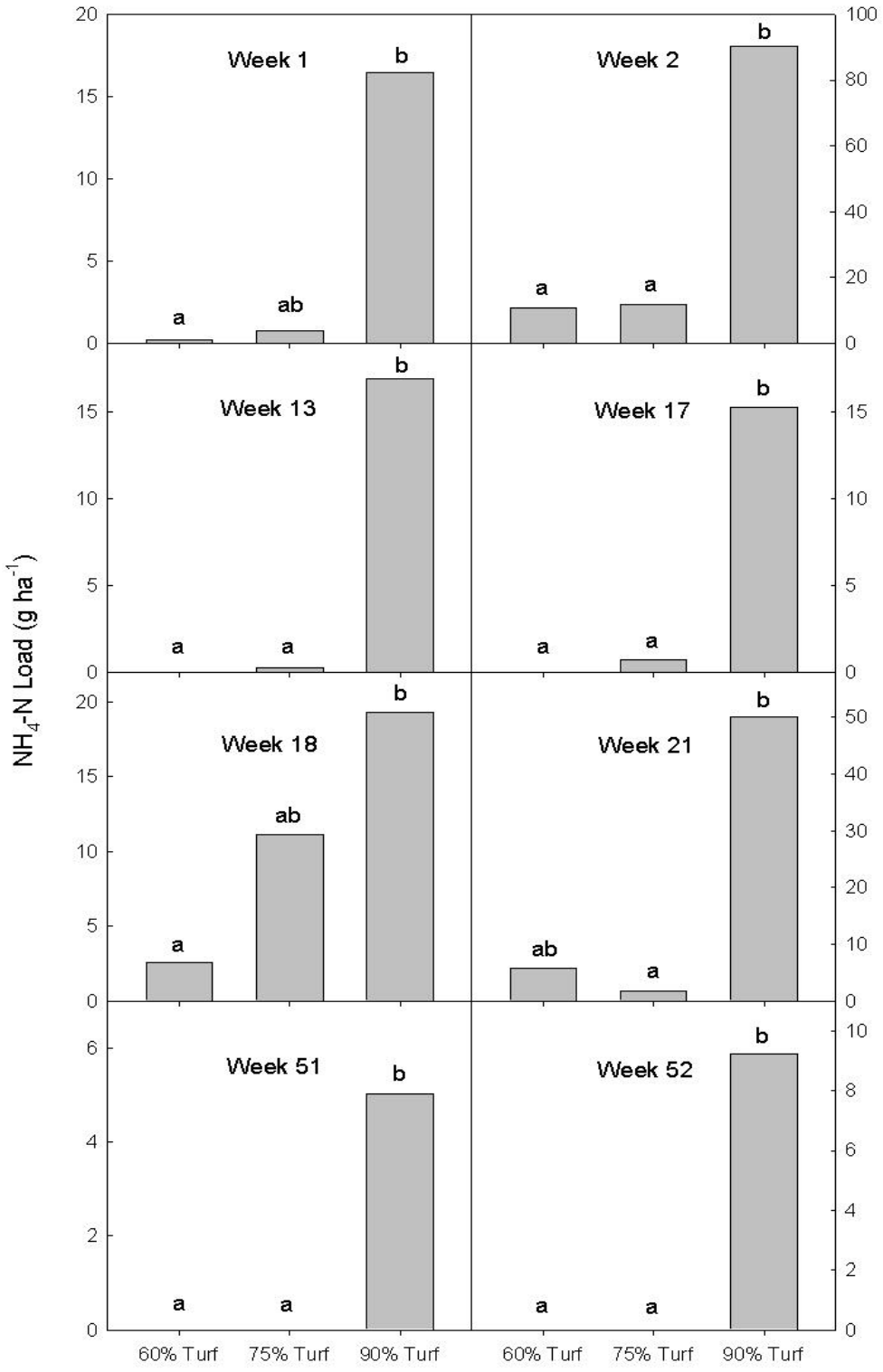

Figure 2-6. Mean weekly NH_4–N loads from lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10) during weeks with a significant plant cover treatment effect.

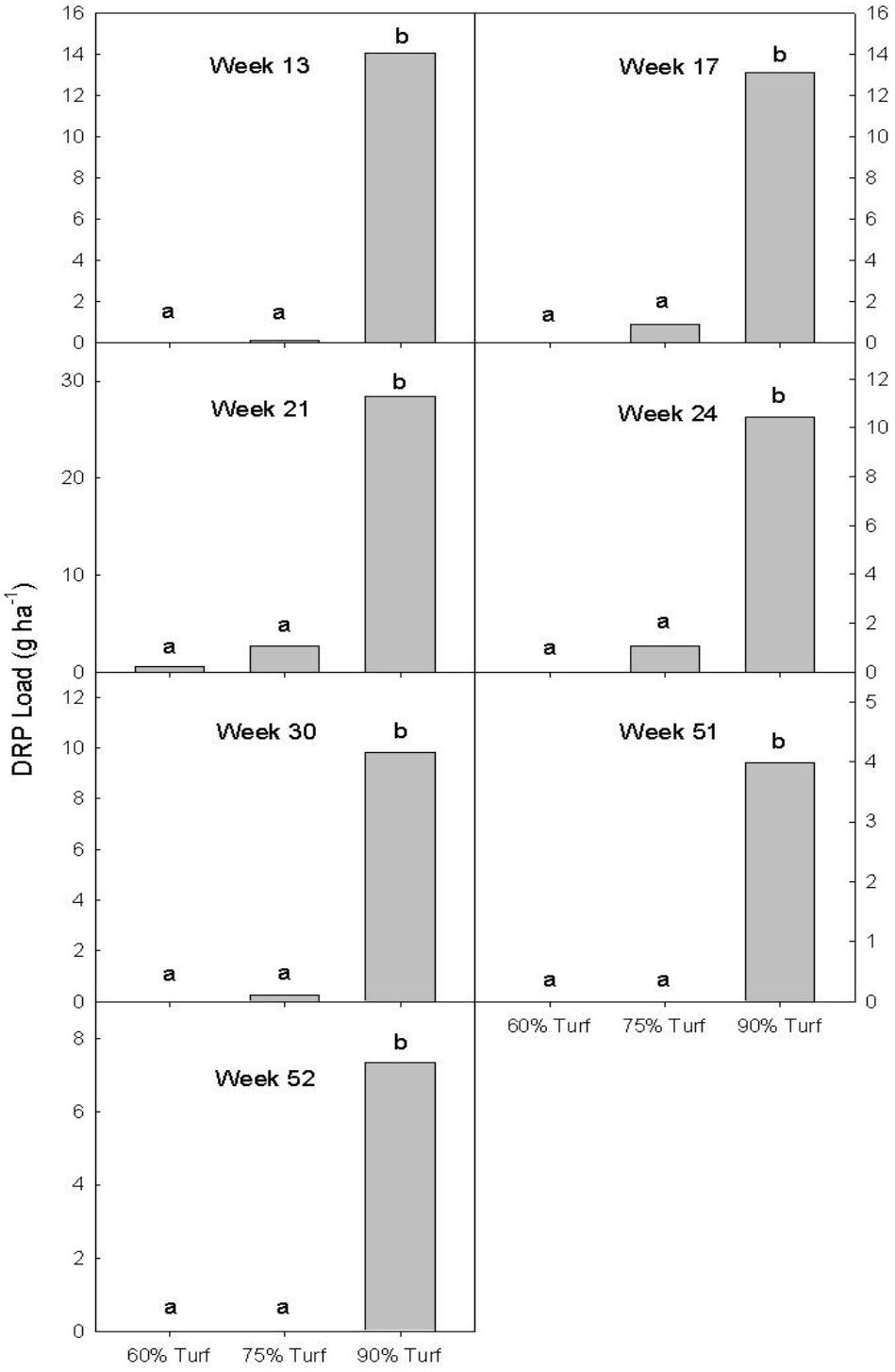

Figure 2-7. Mean weekly dissolved reactive P(DRP) loads from lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10) during weeks with a significant plant cover treatment effect.

CHAPTER 3
A NUTRIENT BUDGET APPROACH TO DETERMINING NITROGEN AND PHOSPHORUS USE EFFICIENCY AND LOSSES FROM MIXED SPECIES RESIDENTIAL LANDSCAPES

Introduction

In recent years, the population growth and development of urban areas has led to non-point source (NPS) pollution of surface and groundwater (Fisher et al., 2006; Hauxwell et al., 2001). Researchers have shown that urbanization processes and other human activities in urban areas can alter the natural vegetation cover, hydrology cycle, and nutrient fluxes in urban systems (Roach et al., 2008). Human-based activities have changed the physical and chemical characteristics of urban soils when compared with soil properties before disturbance (Jim, 1998). Soil characteristics that were commonly reported for urban soils include increased soil pH and bulk density (Gregory et al., 2006; Jim, 1998), degraded soil structure, low organic matter content and nutrient storage capacity (Law et al., 2004), low water holding capacity, and low biological activity, which often indicated low soil quality. As a result, frequent inputs of irrigation and fertilizers are often required to overcome plant growth and quality issues related to low soil quality in urban soils. However, fertilizers applied in urban landscapes can be lost to the environment (e.g., runoff and leaching) if nutrients are not absorbed by plants or stored in soils (Petrovic, 1990; Soldat and Petrovic, 2008). The loss of applied nutrients to the environment reduces the availability of nutrients for growing plants and increases the potential risk of water quality degradation. Thus, the development of best management practices (BMPs) that target at urban areas are needed to control the nutrient losses from urban soils.

Nutrient budgets are a valuable management tool that can identify areas with nutrient surplus and guide the development BMPs to control point and non-point source pollution in agriculture (Oenema et al., 2003; Watson et al., 2002). Nutrient budget analysis is defined as the comparison of nutrient inputs (fertilizers, manures, biological N_2 fixation, etc.) and outputs (crop harvest, manure relocation, etc.) in a specific geographic area (Oenema et al., 2003; Slaton et al., 2004; Watson et al., 2002). A positive nutrient budget (inputs > outputs) suggests that nutrients would accumulate in nutrient sinks (e.g., soil, water bodies, biomass, etc.), resulting in an increasing risk of nutrient losses. In contrast, a negative nutrient budget (inputs < outputs) suggests that stored nutrients in sinks will be depleted with time gradually. For example, a negative soil nutrient budget suggests that soil nutrients are decreasing and adequate concentrations may not be available to support optimum plant growth in subsequent years (Oenema et al., 2003). Nutrient surpluses should be prevented by increasing nutrient use efficiency (i.e., increase the proportion of applied nutrient to be absorbed by plants and/or minimizing the off-site movement of nutrients from the soil into water or air). Preventing nutrient deficits will maintain sufficient levels of nutrients (mainly N and P) for plant uptake. The most ideal nutrient management situation in agricultural and urban landscape is a balanced nutrient budget (inputs = outputs).

During the past several decades, nutrient budgets were used widely by farmers to guide nutrient management BMP implementation (Oenema et al., 2003) and by policy makers as a tool to develop regulatory policies to control nutrient pollution originating from agricultural systems (DeWalle and Sevenster, 1998). Nutrient budgets were used in agricultural systems at various geographical scales, ranging from farms (Buciene et

al., 2003), states (Lanyon et al., 2006; Slaton et al., 2004), regions (Sacco et al., 2003), and countries (Lord et al., 2002). For example, Tunney et al. (2003) summarized soil surface P balance conditions by sending questionnaires to representatives in 18 countries that participated in European Union's European Cooperation in the Field of Scientific and Technical Research action on "Quantifying the Agricultural Contribution to Eutrophication (COST 832)". The results showed that the net P balance varied among countries, suggesting differences in soil fertility conditions and risk of nutrient losses from agricultural sources. In general, reduced soil fertilizer inputs led to negative soil surface P balances in several eastern European countries (e.g., Hungary and Poland), which may gradually lead to a depletion of soil P storage. However, applied P fertilizer continued to accumulate in soils (positive soil surface P balance) in western European countries despite reduced fertilizer inputs in recent years. These results suggested that soils in many western European countries had high risk of nutrient losses (Tunney et al., 2003). At a much smaller geographical scale, Obour et al. (2011) conducted a soil surface P budget for bahiagrass pastures system growing on a Florida Spodosol and was fertilized at three application rates (0, 5, and 10 kg ha^{-1} P). When P stored in Bh horizon was included in the calculated P budget, the net P balance was positive for all fertilizer treatments, including the unfertilized control. Based on the positive P budgets, the authors suggested that P stored in the Bh horizon was a significant P source for plant uptake; therefore, P fertilizer inputs to pastures growing on Spodosol in Florida should be reduced (Obour et al., 2011). These examples demonstrate the potential of nutrient budgets to identifying areas with nutrient surplus or deficit in agriculture

systems. Nutrient balance information can be used in nutrient management to maximize nutrient use efficiency and reduce the risk of nutrient losses to the environment.

However, few studies evaluated the use of nutrient budgets in urban areas (Baker et al., 2001; Watson et al., 1981) when compared to the number of studies conducted in agricultural areas. The use of nutrient budgets in urban areas has typically been conducted at larger scales (e.g., watershed, region). For example, Watson et al. (1981) calculated a nutrient budget (N and P) for Lake Wingra watershed in Wisconsin (USA) (approximately 75% urban and 25% natural wooded lands) to determine the impact of urbanization on the watershed. The results indicated that surface runoff was the dominant form of P inputs to the watershed due to the increased runoff events that resulted from urbanization. As a result, the ratio of N to P in annual loadings to the Lake had decreased due to the significant increase of P loading to watershed via runoff. The authors recommended that water quality managers should work to reduce the movement of P from urban system in runoff (Watson et al., 1981). In another study, Baker et al. (2001) determined that both urban and agricultural (cropland and dairy system) components of the Central Arizona-Phoenix ecosystem had an order of magnitude higher N inputs than natural systems (e.g., desert) using a nutrient budget approach. The human transported N inputs constituted 88% of total N inputs, with food and fertilizer as the dominant sources (Baker et al., 2001).

Moreover, the use of nutrient budget in small urban scales (e.g., lawn, mixed landscape plot) is lacking in literature. The typical urban and residential landscape plots in the U.S. consist of a mixture of turfgrass and ornamental plants such as shade trees and woody shrub species (Hipp et al., 1993). Calculation of nutrient budget in these

small geographical scales (e.g., lawn, mixed landscape plot) can help us understand how much nutrients were used by plants, stored in nutrient sinks (e.g., soils and plants), and lost to environment (e.g., leachate and runoff), which guides the development of BMPs to reduce NPS pollution at lot scale. We can also use this information to identify landscape compositions that maximize nutrient use efficiency and minimize nutrient losses to environment. Therefore, the objective of this study was to calculate nutrient budgets for mixed landscape plots containing varying proportions of turfgrass and woody ornamental species. We hypothesis that less nutrients will be lost and total nutrient budget will be more balanced in landscape with higher proportion of established woody ornamental species.

Materials and Methods

Lysimeter Design

Nine landscape lysimeters (3.35 m wide × 3.96 m long) were constructed at the University of Florida-Institute of Food and Agricultural Sciences (UF-IFAS) Mid Florida Research and Education Center in Apopka, FL. Lysimeters were constructed into a hill side with only the west facing wall fully exposed. Each lysimeter was 1.45 m deep along the outside edge and 1.52 m deep in the middle. The bottom of each lysimeter was sloped towards the center with a single drain pipe exiting the wall for drainage collection. The inside of each lysimeter was painted twice with basement wall waterproof paint. The drainage system was designed around a central junction box over the center drain hole with sock-covered 10.2 cm corrugated drain pipe extending to diagonal corners. Lysimeters were first covered with approximately 61 cm layer of rock, textile cloth, and coarse sand, which was overlain with roughly 80 cm layer of native soil (a mixture of Candler-Apopka and Tavares-Millhopper soil series). In August 2009, soil

in each lysimeter was leveled to about 7.62 cm below the top of each wall in preparation for installation of plant material. Backfill soil was watered-in to insure good contact between root balls and the soil at planting. Excess soil was removed from lysimeters.

Three vegetative cover treatments containing varying proportions of turfgrass and woody ornamental cover were installed in the lysimeters as follows: 1) 90% St. Augustine turfgrass, 10% woody ornamentals; 2) 75% turfgrass, 25% woody ornamentals, and 3) 60% turfgrass, 40% ornamental cover. All treatments contained one *Magnolia grandiflora* L. 'D.D. Blanchard' (magnolia) in the center of lysimeter. The magnolias were transplanted from 50.8-cm root control bags (High Caliper Growing Systems, OK) on 9 Sept. 2009. The magnolias were approximately 1.83 m tall and 3.81 cm in caliper as measured at 15.2 cm above ground. An area of approximately 1 m^2 was mulched around the magnolia. The 75 and 60% turfgrass treatments also contained *Viburnum odoratissimum* Ker Gawl. (sweet viburnum) shrubs. The viburnum shrubs were transplanted from 11.4-L (#3) containers on 9 Sept. 2009. The 75% turfgrass treatment which contained one hedge consisting of two shrubs, mulched 0.91 m wide and 1.98 m long, was placed in the northeast corner of the lysimeters. The 60% turfgrass treatment contained two hedges. *Stenotaphrum secundatum* (Walter) Kuntze 'Floratam' (St. Augustinegrass) sod was cut from a sand soil on 24 Sept. 2009 and was installed to cover the remaining surface area of each lysimeter on 25 Sept. 2009. Turfgrass and woody ornamental species were considered fully established by the end of May 2010.

Irrigation and Rainfall Volume Quantification

An irrigation system was installed in each lysimeter prior to planting. Pop-up spray heads (PROS-06-10A, Hunter Industries, Inc., San Marcos, CA) were positioned at

each corner and in the center along the north and south sides of each lysimeter for irrigation of turfgrass areas. Shrub and tree irrigation was delivered through a 1.91 cm black polyethylene tubing trunk line outfitted with a 172 kPa pressure regulator. The tree was irrigated using two 30.5 cm tree stakes with 189 L h^{-1} nozzles and an inverted cone spreader (Jain Irrigation Inc., Fresno, CA). Shrubs were irrigated with the same spray stake assembly using 102 L h^{-1} nozzles (Jain Irrigation Inc., Fresno, CA). One stake was placed between each plant and along both outside edges of the lysimeter wall. Irrigation valves were positioned on the outside of the west-facing wall of the lysimeter. One water meter (C700-SF, Elster-Amco, Ocala, FL) with an electronic counter (32 counts L^{-1}) was installed above each valve to measure volume of irrigation applied.

The irrigation applied to plants followed a model that took into consideration of evapotranspiration (ET_o) and the projected surface area of the plant canopy (Beeson, 2005).Woody species were irrigated using micro-irrigation on alternate days after transplanting until late May 2010. Beginning the 1 June 2010, turfgrass and ornamental irrigation was controlled using a CR10X data logger program (Campbell Scientific, Inc, Logan, UT). Reference ET_o and daily rainfall amount were calculated at midnight by the onsite weather station and transferred to the CR10X data logger controlling the system. The weather station consisted of a LI200X pyranometer (Li-Cor Inc., Lincoln, NE), a CS215 temperature and relative humidity probe (Campbell Scientific, Inc., Logan, UT), a wind sentry set (03001, R.M. Young Co., Traverse City, MI), and a tipping bucket rain gauge (TE525, Texas Electronics, Dallas, TX) connected to the CR10X data logger. Irrigation depths were calculated by multiplying ET_o by the plant coefficient (Kp) for each species (0.73, 0.70, and 0.90 for magnolia, viburnum, and turfgrass, respectively

derived from previous experiments) and the ratio of the horizontal planar projected canopy area (PCA) of each species (magnolia, viburnum, and turfgrass) relative to the total surface area of each lysimeter. The Kp values were determined based on the irrigation frequency that provided an acceptable level of aesthetic quality (i.e., no brown and curling leaves, minimize drought stress) for each species (R. Beeson and M. Dukes, personal communication, 2011) and the PCA was determined every 3 weeks and updated in the data logger program. Turfgrass irrigation began at 0500 h to minimize the time turfgrass was wet to reduce disease pressure and achieve better irrigation uniformity. Shrub and tree irrigation was initiated at 0700 h.

The irrigation depths were summed each day and added to a running total irrigation depth for each lysimeter. When the cumulative total exceeded 19.0 mm of irrigation, the irrigation depth was multiplied by the area of the turfgrass to calculate the volume of water applied for the turfgrass. The PCA of the magnolia and viburnums were multiplied independently by the cumulated irrigation depth and the totals were added and the volume of irrigation was applied equally to each species. Total irrigation volume for the magnolia and viburnums defaulted to 19 L (never happened during our study period) when volumes were smaller.

Fertilization

Fertilizers were applied initially to the plots on 11 Mar. 2011 following UF-IFAS recommendation for basic St. Augustinegrass maintenance in central Florida (Sartain, 2007).The fertilizer rates applied to woody ornamental beds were production level rates to accelerate growth from newly installed beds to mature landscape in three years. Turfgrass areas received a received a granular complete fertilizer (‘Natural Green’, 8-2-8, ProSource One, Lake Alfred, FL) at a rate of 60 g m^{-2}. A controlled-release complete

fertilizer (18-6-8, 180 d release, Florikan, Sarasota, FL) was applied to the magnolia and each viburnum hedge at rates of 220 and 244 g m^{-2}, respectively. A water soluble complete fertilizer ('Magic Carpet', 16-4-8, ProSource One, Lake Alfred, FL) was applied to the magnolia and each viburnum hedge at rates of 14.0 and 13.3 g m^{-2}, respectively. A second application of fertilizers was applied at the same rates to both turfgrass and woody ornamental plants on 12 Sept. 2011.

Soil Sampling and Analysis

Soil samples were collected at the start of the project (7 Mar. 2011) and then every three months (until December 2011) at a depth of 0 to 15 cm using a soil probe sampler. Each lysimeter was gridded into 16 units (approximately 0.83 m^2 $unit^{-1}$) and a soil sample was collected from each unit. Soil samples from turfgrass areas and woody ornamental areas were collected and composited separately to create two soil samples per lysimeter. Composite samples were air-dried at 25 ± 2 ºC and sieved through 2-mm screen before analysis. Soil test P, Al, and Fe were analyzed using inductively coupled plasma – atomic emission spectroscopy (ICP-AES) (USEPA Method 6010C) after Mehlich-3 extraction (Sims, 2009).

Plant Tissue Sampling and Analysis

Turfgrass was mowed when growth exceeded 5.7 cm and clippings were collected. Shrubs growth that exceeded a height of 1.67 m, a width (north/south) of 0.92 m, and a length (east/west) of 1.98 m (75% turfgrass treatment) or 3.96 m (60% turfgrass treatment) was removed. Turfgrass and shrub leaf tissue samples were oven dried at 105 ± 2 ºC and weighed. Plant tissue was then ground using a Wiley mill (Arthur H. Tomas Co., Swedesboro, NJ) and digested for total Kjeldahl N (TKN) and total P using the standard method of the UF-IFAS Extension Soil Testing Laboratory

(Mylavarapu, 2009). Plant tissue TKN and total P were analyzed by autoanalyzer and ICP-AES, respectively, after digestion.

Daily Leachate Volume Quantification

A dry well was installed below the 5.08-cm lysimeter drain, which consisted of an upper collection vessel with a volume of 1.5 L that drained into a weighing vessel through a normally open valve. The weighing vessel was suspended from a 22.7 kg load cell (Interface Inc, Scottsdale, AZ) with a normally closed valve in the bottom. The weighing vessel also had an overflow drain installed near the top to channel water below the drain valve should leachate exceed the capacity of the system. When a computer determined the vessel should be drained, a 12 volt (direct current) sump pump was activated to close the upper valve (normally open) and stop the flow into the weighing vessel; concurrently, the drain valve (normally closed bottom valve) was opened to evacuate the weighing vessel. Once empty, the pump power was turned off and the leachate that had drained from the lysimeter during the evacuating process was allowed to proceed to the weighing vessel. The measuring device was located inside an enclosure with a tin roof and cement board sides (Hardie board, James Hardie, Mission Viejo, CA) that were sealed with silicon and expanding foam to exclude rainfall, dust, and blowing sand.

The system determined the mass of each weighing vessel every 2 min and compared the mass to the empty vessel. When a minimum of 1 L of leachate was collected, the measuring device valves were activated and the measuring vessel was then weighed every 10 s. When all water drained out, the valves were de-activated and the amount drained was added to a running daily total. The volume of drainage water from each lysimeter was recorded daily at 0500 h, when the running total was stored by

a computer. The leachate volume was then reset to zero and daily volume determination began again.

Leachate Sampling and Water Sample Analysis

Drainage water from the nine lysimeters containing established mixed landscapes was collected from 5 Mar. 2011 to 9 Mar. 2012. Water drained through the lower measuring valve flowed horizontally through 10-cm of braded polyurethane tubing (12.5 mm inner diameter) and then flowed vertically out of the tube through a 90 degree polyethylene elbow. Leachate was sampled by inserting a 16G 1 needle (Becton Dickinson Co., Rutherford, NJ) through the bottom of the polyurethane tubing. As water flowed through the tubing, 1.0 ± 0.2% of the total drainage water passed through the needle and collected in an open top 3.0-L graduated beaker. Leachate collected in these beakers was retrieved daily at 0745 h (Monday through Friday) to serve as the daily subsample. Weekend subsamples were collected the following Monday. Daily leachate subsamples were stored frozen until they were combined to produce a weekly flow-weighted composite sample (based on total volume).

Irrigation, rainfall, and leachate samples were filtered through a 0.45-μm filter and then split into equal portions for analysis of N and P. One sample was digested using modified USEPA Method 351.2 (USEPA, 1993b) and analyzed for TKN using a discrete analyzer (AQ2, Seal Analytical, West Sussex, UK). The modified method employed a 5 to 1 sample to solution ratio instead of the 2.5 to 1 sample to solution ratio outlined in USEPA Method 351.2. Another sample portion was analyzed for NO_3 + NO_2–N (USEPA, 1993c) colorimetrically using a discrete analyzer (AQ2, Seal Analytical, West Sussex, UK). The last portion of leachate was analyzed using molybdate-blue reaction to determine dissolved reactive P (DRP) (Murphy and Riley, 1962). Nutrient loads were

calculated by multiplying the total leachate volume by the nutrient concentration. N loads in our study were calculated by combining TKN load and load of NO_3+NO_2–N

Calculation of Lysimeter Nutrient Budgets

A soil surface budget method was used to calculate the net nutrient (N and P) balance within each lysimeter. The nutrient budgets (N and P) were calculated separately for each lysimeter in each quarter (Q): Q1, 5 Mar. 2011 to 3 June 2011; Q2, 4 June 2011 to 2 Sept. 2011; Q3, 3 Sept. 2011 to 2 Dec. 2011; and Q4, 3 Dec. 2011 to 9 Mar. 2012. An annual nutrient budget (N and P) was also calculated for the whole study period (5 Mar. 2011 to 9 Mar. 2012). Nutrient inputs, outputs, and storage were considered when calculating the soil surface budget.

The nutrient (N and P) budget for each lysimeter was calculated using the following equation (Obour et al., 2011): Net nutrient balance (kg ha^{-1}) = (F_i + I_i + R_i+ S_s) – (U_o + L_o),where F_i = fertilizer input (kg ha^{-1}), I_i = irrigation input (kg ha^{-1}), R_i = rainfall input (kg ha^{-1}), S_s = soil stored plant available nutrient (kg ha^{-1}), U_o= plant nutrient removal (kg ha^{-1}), L_o= leached nutrients (kg ha^{-1}). Soil stored plant available nutrient (Mehlich 3 P) was only considered in the P budget because studies showed that soils can potentially be a significant source of P available for plant uptake (Obour et al., 2011; Oyanarte et al., 1997). Soil stored N was not considered in our study because plant available N (mainly NO_3^-) is prone to movement in soil due to its high solubility (Pathak et al., 2004) and its negative charge (repelled by soil exchange sites), which limit the accuracy of tested plant available N in soils over time under the condition of infrequent soil sampling (every three months). Unlike typical soil surface budget that ignores potential loss of nutrients to environment (Obour et al., 2011), we considered nutrient leaching in our budgets.

Fertilizer input (F_i) was determined by multiplying fertilization rate by the area of the lysimeter. Nutrient inputs from irrigation (I_i) and rainfall (R_i) were calculated by multiplying the nutrient concentration in irrigation and rainfall samples with total volume of irrigation and rainfall applied, respectively. Rainfall volumes applied were assumed to be the same for all lysimeters for each precipitation event. Soil mass was assumed to be 2,250,000 kg ha^{-1} to a depth of 15 cm. Soil mass under turfgrass and woody ornamental bed areas were estimated by multiplying the percentage of turfgrass and woody ornamental plants with total soil mass in each lysimeter, respectively. Plant available P mass was calculated by multiplying the Mehlich 3 P concentration by the soil mass for both turfgrass and woody ornamental areas. Total soil stored available P (S_s) for each lysimeter was the sum available P mass from turfgrass and woody ornamental plants. Annual soil P stored was estimated by taking the average of the soil P storage for all four quarters. Plant nutrient removal (U_o) was calculated by multiplying nutrient concentration in tissue samples by the dry weight of clippings. Nutrient concentrations of irrigation, rainfall, soil, and plant tissue were assumed to remain constant between sampling dates. Nutrient output in leachate (L_o) was calculated as nutrient loads (nutrient concentration × leachate volume).

Statistical Analysis

The experiment was designed as a randomized complete block design with three vegetation treatments applied in three blocks (replicates) (each block consisting of a set of three consecutive lysimeters). Parameters were log transferred before analysis. All tested parameters (except rainfall volume and its nutrient inputs, and fertilizers) were analyzed using the PROC MIXED procedure by SAS with quarter as fixed effect and block as random effect to determine temporal trends(SAS Institute, 2008).All tested

parameters (except rainfall volume and its nutrient inputs, and fertilizers) were also analyzed using the PROC MIXED procedure of SAS (by quarter or annually) with treatment as fixed effect and block as random effect to determine the vegetation cover effect (SAS Institute, 2008). Normality was checked by examining histogram and normality plots of the conditional residuals. All pairwise comparisons were completed using the Tukey's honestly significant difference test with a significance level of $\alpha = 0.05$.

Results and Discussion

Nutrients Inputs

In our study, fertilizer was the dominant nutrient source in both N and P budgets, accounting for 95 to 98% of total applied N and 97 to 99% of total applied P over the entire study period (annual), depending on vegetation treatment. Fertilizers were not applied during summer months (Q2), when intense precipitation occurs to minimize NPS nutrient losses. When fertilizers were applied, higher amount of total N (Table 3-1) and P (Table 3-2) were applied to treatments containing higher proportion of woody ornamental plants (60% turfgrass> 75% turfgrass>90% turfgrass).

Rainfall N (Table 3-1) and P (Table 3-2) inputs generally followed rainfall distribution patterns with the least rainfall nutrients inputs in Q4 (winter) due to the low precipitation event (Table 3-3). The only exception was noted in Q1, where rainfall volume was highest (Table 3-3), but P rainfall input was low (Table 3-2). This is mainly because of the very low DRP concentration in rainfall samples (mean DRP concentration was 0.03 mg L^{-1}) during Q1 period. Historical data indicated that South-Central Florida Ridge areas (parts of the east side of the city of Tampa Bay and the west half of Orlando are included) typically have most of the rainfall from late spring

through early autumn and were relatively dry in late autumn and winter section (USDA-NRCS, 2006), which corresponded to our low rainfall nutrient inputs in winter quarter (Q4).

In our study, irrigation nutrient inputs were basically determined by irrigation water input due to the low nutrient concentrations in irrigation samples (maximum N (TKN+NO_x) and DRP concentrations were 2.91 and 0.16 mg L^{-1}, respectively).In general, less irrigation was applied in Q3 and Q4 for all three treatments (Table 3-4) mainly because of the lower temperature in these periods decreased the ET_0 of the plants (Kosa et al., 2006) compared with in Q1 and Q2. Thus, less N and P in irrigation were added to lysimeters in these two quarters (Table 3-5). Moreover, significantly higher irrigation volume was applied to the 90% turfgrass treatment compared to the 60 and/or 75% turfgrass treatment in all four quarters (Table 3-3). Treatment containing 75% turfgrass also received higher irrigation water than 60% turfgrass treatment in Q1and Q4 (Table 3-3). Annual irrigation volume increased with increasing turfgrass proportion in landscape (90% turfgrass >75 turfgrass > 60% turfgrass) (Table 3-3). This was mainly because we used a higher Kp value for turfgrass (0.9) than woody ornamental plants (0.73 and 0.70 for magnolia and viburnum respectively) when determining the irrigation depth. The lower irrigation inputs to treatments containing higher proportion of woody ornamental plants were also consistent with the fact that ornamental plants generally require less irrigation than turfgrass to maintain a similar visual quality level by extending deep root system to use stored soil moisture (Pittenger et al., 2001). As a result, significantly higher N (Table 3-1) and P (Table 3-2) were applied to 90% turfgrass treatment than 60 and/or 75% turfgrass treatment in every quarter. Annual irrigation

nutrients inputs also increased with increasing turfgrass proportion in landscape (90% turfgrass >75% turfgrass > 60% turfgrass) (Table 3-1 and 3-2).

Mehlich 3 Soil P Storage

Our soil P test results revealed that plant available P (Mehlich 3 extracted) in soil surface (0 to 15 cm) storage was a substantial source of P for plant uptake, which equated to approximately three to nine times of total applied P fertilizer depending on plant cover treatment. Obour et al. (2011) indicated that P (Mehlich 1 extracted) stored in the surface 15 cm soils was approximately 1.1 to1.5 times of applied fertilizer P depending on application rate (5 or 10 kg ha^{-1}) in a typical Florida Spodosol. However, P stored in the surface 45 cm soils (including Bh horizon) was 9.5 to 18.3 times of total applied P, suggesting a great storage of plant available P in soils. Also, after converting initial Mehlich 3 P concentrations (Table 3-6) in our soil to comparable Mehlich 1 concentrations (Mylavarapu et al., 2002), soil interpretation indicated that the P level in our soils under turfgrass was medium, in which case the probability of plant response to applied fertilization would be ≤25% (Sartain, 2008). This indicated that our soil storage of P was already relatively high before fertilization events.

Plant cover did not affect the plant available P storage in soil (Table 3-2), but sampling date did affect soil P content (Table 3-5). In all three treatments, plant available P storage increased significantly from Q1 to Q3, and leveled off in Q4 (Table 3-5). But the higher soil P content in Q3 than in Q1 may just due to the timing of sampling event. Soil samples collected in Q3 were taken (15 Sept. 2011) right after fertilization event (12 Sept. 2011), while soil samples in Q1 (7 Mar. 2011) were taken prior to fertilization (11 Mar. 2011).Thus, we consider soil P content was generally stable in our study; as we can see that no significant differences were noted between

Q2 and Q4 (quarters without fertilization) in all treatments on soil P content (Table 3-5).This may due to the initially high Fe and Al concentrations in our soils (Table 3-6), which form strong chemical bonds with P (Borggaard et al., 2005; Tunesi et al., 1999) and subsequently limit the P mobility in soils. For example, Kleinman and Sharpley (2002) reported that the highest concentration of Al and Fe of 37 acidic and 25 alkaline surface soils obtained from USDA Natural Resources Conservation Service's National Soil Survey Laboratory was 1.43 and 0.41 mg kg^{-1} respectively, which was lower than the concentration noted in our soils (Table 3-6). Another possible explanation for the stable soil P content over time is that our soils already used up the majority of their capacity to absorbing P at the beginning. Nair et al. (2004) reported that soil water soluble P and P loss potential increased significantly when the degree of P saturation (DPS) ((M3-P)/α(M3-Fe+M3-Al))×100, where M3-P, M3-Fe and M3-Al is the concentration of Mehlich-3 extractable P, Al, and Fe expressed in mg kg^{-1}, α is an empirical factor considering P saturations of different soils) was above a threshold of 16% for Florida sandy soil. The DPS values of our soils (38 and 27% for soils under turfgrass and ornamental plants respectively) were higher than the reported threshold by Nair et al. (2004), suggesting that our soils had low ability to hold additional P and applied fertilizer P would be less likely to accumulate in soil storage.

Plants Nutrients Removal

Plant nutrient removal was the main form of nutrients output from the landscape lysimeters, accounting for, on average, 86 and 98% of annual total N and P outputs, respectively. Plant nutrient removal (both N and P) was affected by both quarter and vegetation treatment. The highest plant N and P removal was reported in Q2 for all three plant cover treatments (Table 3-5), which corresponded with the summer growing

season that was dominated by frequent precipitations and warm temperature that stimulate active growth of both turfgrass and ornamental plants.

A plant cover treatment effect was reported for plant N removal only in Q3, where plant N removal from lysimeters planted with the 75% turfgrass treatment was higher than plant N removal from than 60 and 90% turfgrass treatments (Table 3-1). However, the annual N budget indicated that there was no significant difference of vegetative treatment on plant N removal (Table 3-1), but 90% turfgrass treatment removed highest percentage total applied N in clippings, followed by 75% turfgrass treatment and then 60% turfgrass treatment (Table 3-7). Similarly, plant P removal from lysimeters planted with the 75% turfgrass was significantly higher than from lysimeters planted with the 60% treatment in only Q3 (Table 3-2). Annual nutrient budgets indicated that removal of P in clippings from the 90% turfgrass treatment was significantly higher than the amount of P removed in tissue from the 60% turfgrass treatment (Table 3-2). Also, tissue P removal from the 90% turfgrass treatment accounted for the highest percentage of total applied P, followed by 75% turfgrass treatment and the 60% turfgrass treatment (Table 3-7).Higher percentages of applied nutrients (both N and P) removed in plant tissue from the treatment containing the highest proportion of turfgrass was probably due to the removal of more aboveground biomass from turfgrass (0.33 kg m^{-2} in average for the study period) when compared with woody ornamentals plant (0.20 kg m^{-2} in average for the study period) and higher fertilizers input to treatments containing higher proportion of woody ornamental plants. It is worth mention here that majority large proportion of aboveground turfgrass biomass was removed from landscape during mowing event. In contrast, only a small proportion of woody ornamental plant tissue (just leaf) was

removed offsite. More nutrients would be stored in treatments containing higher proportion of woody ornamental plants since the established ornamental plants produce higher root (Jackson et al., 1996) and aboveground biomass than turfgrass, which potentially trapped more nutrients in plant body storage onsite.

Leachate Volume and Nutrient Loads

Leachate volumes from lysimeters were consistent with the distribution of rainfall, with the smallest leachate volumes collected during the driest quarter (Q4) for all treatments (Table 3-3 and 3-4). In Q2 through Q4, significantly higher leachate volumes were noted in 90% turfgrass treatment than 60 and/or 75% turfgrass treatment (Table 3-3). However, no vegetation treatment effect was noted in Q1 and for the annual volume of leachate, but leachate volume increased numerically with increasing proportion of turfgrass (Table 3-3). This may because that annual leachate volume was also affected by irrigation in addition to rainfall. Theoretically, the computer-controlled irrigation condition would apply the exactly same irrigation to lysimeters with the same treatment. However, our study was a field experiment with just three replicates. As a result, irrigation received by each replicate may vary depending on micro-climate, plant growing condition and other factors that affect each lysimeter separately, which may generate relatively high variability among replicates and mask the vegetation effect on annual leachate volume. However, 90% turfgrass treatment had significantly higher annual cumulative leachate volume than 60% turfgrass treatment (P value = 0.05) with a significant level $\alpha=0.1$.

Vegetation treatment effect affected leached N load in our study. In Q1 through Q3, leachate N loads exported from 90% turfgrass treatment was significantly higher than the mass of N leached from lysimeters containing the 60 and/or75% turfgrass

treatments (Table 3-1). No plant cover treatment effect was noted in Q4 (Table 3-1), which was likely a result of low precipitation event during this quarter (Table 3-3). Annual nutrient budgets indicated that the 90% turfgrass treatment leached more total N mass (Table 3-1) and higher percentages of total applied N in leachate (Table 3-7) than the 60 and 75% turfgrass treatments.

Plant cover treatment also affected leachate P load during our study. In Q2 and Q3, lysimeters containing the 90% turfgrass treatment leached a significantly higher mass of P than the 60 and/or 75% turfgrass treatment; no plant cover treatment effect was observed in Q1 and Q4 (Table 3-2). There was no plant cover treatment effect on annual mass of leached P, but leached P mass increased numerically with increasing proportion of turfgrass (Table 3-2). Similarly to leachate volume, 90% turfgrass treatment had significantly higher annual DRP load exported than 60% turfgrass treatment (P value= 0.08) with a significant level α=0.1. Moreover, significantly higher percentage of total applied P lost in leachate from 90% turfgrass treatment than 60 and 75% turfgrass treatments annually (Table 3-7).

Our results differed from those of Erickson et al. (2001), who showed higher annual drainage volume (2,237 mm) and NO_3–N loads (48.3 kg ha^{-1})in leachate from mixed species landscapes (mainly woody ornamentals) than from St. Augustinegrass monoculture (2,082 mm and 4.13 kg ha^{-1}) in the first year after planting. Cisar et al. (2004) also reported that significantly higher inorganic N (NO_3–N+NH_4–N) mass was leached from woody ornamental plants (annual mean leached inorganic N mass of 4.89 g m^{-2}) than from turfgrass (annul mean leached N mass of 0.41 g m^{-2}). Similarly, Loper et al. (2012) reported higher drainage volume and nutrient (NH_4–N, NO_3+NO_2–N, and

DRP) loads leached from newly planted woody ornamental plant beds than from turfgrass areas.

The difference between our results and results from other studies (Cisar et al., 2004; Erickson et al., 2001; Loper et al., 2012) on leachate volume and nutrient loads from woody ornamental species and turfgrass was likely due to the fact that previous studies measured nutrient export before the plants were fully established (less than one year after planting). Erickson et al. (2008) showed that the root biomass and density were lower in establishing mixed woody ornamental species (100 g m^{-2}) than in turfgrass (450 g m^{-2}), which reduced the ability of plants to remove water and nutrients from the soil compared with established plants. In our study, all of the three species were approximately 1.5 years old when we begin to collect the leachate samples. The root systems of the woody ornamental plants would have increased significantly (may require 6 to 12 months after installation to become fully established) in root spreading and biomass (Gilman, 1990; Wiese et al., 2009) compared with plants in establishing period, which reduces nutrient leaching potential. Moreover, the established deep-rooted woody ornamental plants can cooperated efficiently with shallow rooted turfgrass to absorb water and nutrients from different depths of the soils (Nair and Graetz, 2004; Nair et al., 2007), which would also explain the lower leachate volume and nutrient (both N and P) losses from landscape treatments containing a higher proportion of woody ornamental species.

Nutrients Budgets

During periods when fertilizers were applied (Q1 and Q3), the net N balances were positive (nutrient surplus) and were significantly higher than balances in Q2 and Q4, when the net N balances were negative (nutrient deficient) (Table 3-5). Although soil N

was not tested in our study, we suspect that if we accounted for soil available N storage, the net N balance would remain positive in seasons without fertilizer inputs because soils can store a significant amount of N (Kaye et al., 2008). Kaye et al. (2008) estimated that approximately 100, 21, and 20 Gg total N were stored in the top 40 cm of soils of urban xeric yard, urban mesic yard, and urban nonresidential areas, respectively within and around the Phoenix, AZ (USA) metropolitan area. However, the study by Kaye et al. (2008) was conducted in relatively dry areas with low precipitation amounts. Nitrogen storage in our Florida soils should be lower than that reported by Kaye et al. (2008), since intense precipitation can leach significant amount of N below the root zone (Erickson et al., 2010).

A plant cover treatment effect was reported for the net N balance in Q1 through Q3, where significantly lower net N balances were noted for 90% turfgrass treatment than the 60 and/or 75% turfgrass treatments (Table 3-1). However, no plant cover effect was noted in Q4. Annual N budget indicated that 60% turfgrass treatment had the highest net N balance, followed by 75% turfgrass treatment, and then 90% turfgrass treatment (Table 3-1).

Quarterly and annual P budgets all indicated positive net P balances (nutrient surplus) (Table 3-2). In general, no plant cover treatment effect was observed for net P balance (Table 3-2). However, when the plant available P stored in soils (S_s) was excluded from the P budget calculation, the net P balance in quarters without fertilizer input (Q2 and Q4) changed from positive (nutrient surplus) to negative (nutrient deficient) (Table 3-2).In Q1 and Q3, net P balance (excluding S_s) was the highest in 60% turfgrass treatment, followed by 75% turfgrass treatment, and then 90% turfgrass

treatment with the least net P balance (Table 3-2). No plant cover treatment effect was noted in Q2 and Q4 (Table 3-2). Annual net P balance (excluding S_s) was also the highest in 60% turfgrass treatment, followed by 75% turfgrass treatment, and then 90% turfgrass treatment (Table 3-2). We can see that the inclusion of soil P storage to P budget calculation masked the treatment effects in each quarter because soil P storage was approximately three to nine times the fertilizer P inputs in our study. This also supports the idea that soil P provides a significant source of nutrients for plant uptake (Obour et al., 2011).

To bring the landscape nutrients budgets closer to nutrient balance, N fertilizer inputs to landscapes should be reduced. Although UF-IFAS fertilization recommendation for turfgrass was followed, the fertilization application rate for woody ornamental plants (magnolia and viburnum: 83.7 and 92.1 g m^{-2}) in our study was much higher than UF-IFAS fertilization recommendations for landscape-grown woody ornamentals (11.2 g m^{-2}) (Kidder et al., 1998). We should also consider spliting the fertilization events, which can subsequently reduce the risk of nutrient losses by frequent fertilization at lower rate when the same amount of fertilizers was applied (Easton and Petrovic, 2004). As a result, applied fertilizer amount should be reduced in Q1 and Q3 when the precipitation amounts were relatively high (Table 3-3), and one fertilization application could be added instead in the driest quarter (Q4) to minimize the chance of applied nutrients being washed out of root zone. Moreover, P free fertilizers should have been applied to turfgrass instead of complete fertilizer (i.e., fertilizers containing N, P and K). This was mainly because initial soil test in our study indicated

that the level of soil P storage for plants uptake was medium, which was considered to be an adequate level for optimum turfgrass growth in Florida (Sartain, 2008).

Contrary to our hypothesis that total nutrient budget will be more balanced in landscape with higher proportion of established woody ornamental species, the 90% turfgrass treatment had the nutrient (N and P) net balances closest to zero (nutrient balance) in our study (Table 3-1 and 3-2). This was mainly because that the woody ornamentals in our study were fertilized at a production level rate, which was far higher than UF-IFAS recommendations for woody ornamentals growing in the landscape. However, treatment containing higher proportion of woody ornamentals still leached significantly lower nutrients than treatment with higher proportion of turfgrass (Table 3-1 and 3-2), suggesting that nutrient use efficiency was reduced with the increasing turfgrass component in mixed landscape. We suspected that the inclusion of more woody ornamental species would further reduce nutrient losses from mixed landscape if we reduce the current fertilization rate. As a result, priority should be given to landscapes containing higher proportion of woody ornamental species when designing and managing urban green areas to reduce NPS pollution and increase nutrient use efficiency.

Overall, the results of this study showed that P fertilization of urban green areas should follow soil test recommendations to prevent adding excess nutrients to landscape since soil P storage may be substantial; while N fertilization should follows UF-IFAS recommendations. The study also suggested that the need to have different fertilizer regimes for ornamental plants and turfgrass due to the differences in plant needs and soil properties under different species. Also, we should make full use of the

plant clippings. For example, researchers showed that turfgrass tissue typically contains 2.0 to 5.0 g kg^{-1} P in dry matter (Guillard and Dest, 2003; Johnson et al., 2003). If we recycle the clippings back into the soil system, the nutrients in tissue may be available to plants again. This subsequently increases nutrients use efficiency and reduces fertilizer inputs in the long term (Qian et al., 2003). Nutrient budgets calculation indicated that plant nutrient removal was the dominant form of nutrient outputs. However, our study was conducted in a sandy Florida soil (low runoff potential) and in a warm weather setting with intense precipitation (suitable for plant growth). Other possible nutrient outputs that are of less importance in our study, such as runoff (Erickson et al., 2001), may be important in other areas containing finer-textured soils. Moreover, our study indicated that urban landscapes containing higher proportion of woody ornamental plants requires less irrigation inputs, exported less nutrient (N and P) loads, and had higher nutrient use efficiency than landscape with higher proportion of turfgrass. Although the landscape containing highest proportion of woody ornamental plants had the most imbalanced nutrient budgets in our study, the nutrient surplus condition can be easily solved by reducing fertilizer rate and splitting fertilization events. As a result, we conclude that the incorporation of more woody ornamental plants in urban landscape may create a more environmental-friendly landscape to reduce water usage and NPS nutrient losses.

Table 3-1. Estimated seasonal and annual N budgets for lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10).

Treatment	Fertilizer N (F_i)	Irrigation N (I_i)	Rainfall N (R_i)	Plant N Removal (U_o)	Leached N (L_o)	Net N balance
	kg N ha^{-1}					
Quarter one (5 Mar. 2011 to 3 June 2011)						
60% Turfgrass	186	0.58 a[a]	1.67	6.35 a	2.83 ab	+[b]180 c
75% Turfgrass	131	0.81 b	1.67	9.05 a	1.70 a	+123 b
90% Turfgrass	76.1	1.18 c	1.67	7.00 a	4.06 b	+68.1 a
Quarter two (4 June 2011 to 2 Sept. 2011)						
60% Turfgrass	0.00	1.03 a	1.12	24.1 a	0.81 a	-22.8 c
75% Turfgrass	0.00	1.24 ab	1.12	27.1 a	1.23 a	-26.0 b
90% Turfgrass	0.00	1.48 b	1.12	29.0 a	4.85 b	-31.3 a
Quarter three (3 Sept. 2011 to 2 Dec. 2011)						
60% Turfgrass	186	0.38 a	1.18	9.20 a	0.91 a	+178 c
75% Turfgrass	131	0.45 a	1.18	12.6 b	1.52 ab	+119 b
90% Turfgrass	76.1	0.64 b	1.18	8.44 a	3.51 b	+66.3 a
Quarter four (3 Dec. 2011 to 9 Mar. 2012)						
60% Turfgrass	0.00	0.39 a	0.17	3.75 a	0.00 a	-3.19 a
75% Turfgrass	0.00	0.51 b	0.17	4.00 a	0.00 a	-3.33 a
90% Turfgrass	0.00	0.78 c	0.17	2.99 a	0.84 a	-2.88 a
Annual N budget (5 Mar. 2011 to 9 Mar. 2012)						
60% Turfgrass	372	2.38 a	4.14	43.4 a	4.56 a	+332 c
75% Turfgrass	262	3.02 b	4.14	52.8 a	4.46 a	+213 b
90% Turfgrass	152	4.08 c	4.14	47.4 a	13.3 b	+100 a

[a]Values within the same test parameter with the same letter are not significantly different at $P < 0.05$ using Tukey's HSD test in the same quarter or annual budget.
[b]+ means nutrient surplus, - means nutrient deficit.

Table 3-2. Estimated seasonal and annual P budgets for lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10).

Treatment	Fertilizer P (F_i)	Irrigation P (I_i)	Rainfall P (R_i)	Mehlich 3 P[b] (S_s)	Plant P Removal (U_o)	Leached P (L_o)	Net P balance	Net P balance[d]
	kg P ha^{-1}							
Quarter one (5 Mar. 2011 to 3 June 2011)								
60% Turfgrass	26.0	0.04 a[a]	0.03	117 a	1.14 a	0.08 a	+[c]142 a	+24.9 c
75% Turfgrass	17.7	0.05 b	0.03	120 a	1.80 a	0.10 a	+135 a	+15.9 b
90% Turfgrass	9.50	0.07 c	0.03	134 a	2.10 a	0.13 a	+142 a	+7.37 a
Quarter two (4 June 2011 to 2 Sept. 2011)								
60% Turfgrass	0.00	0.05 a	0.08	139 a	6.45 a	0.03 a	+132 a	-6.35 a
75% Turfgrass	0.00	0.07 ab	0.08	155 a	7.76 a	0.03 a	+147 a	-7.64 a
90% Turfgrass	0.00	0.08 b	0.08	167 a	9.12 a	0.12 b	+158 a	-9.08 a
Quarter three (3 Sept. 2011 to 2 Dec. 2011)								
60% Turfgrass	26.0	0.03 a	0.08	167 a	3.07 a	0.03 a	+191 a	+23.0 c
75% Turfgrass	17.7	0.04 a	0.08	197 a	4.11 b	0.05 ab	+210 a	+13.7 b
90% Turfgrass	9.50	0.05 b	0.08	182 a	3.93 ab	0.11 b	+187 a	+5.59 a
Quarter four (3 Dec. 2011 to 9 Mar. 2012)								
60% Turfgrass	0.00	0.03 a	0.03	148 a	0.93 a	0.00 a	+147 a	-0.87 a
75% Turfgrass	0.00	0.05 b	0.03	176 a	1.21 a	0.00 a	+176 a	-1.13 a
90% Turfgrass	0.00	0.07 c	0.03	178 a	1.40 a	0.04 a	+176 a	-1.34 a
Annual P budget (5 Mar. 2011 to 9 Mar. 2012)								
60% Turfgrass	52.0	0.16 a[a]	0.22	142 a	11.6 a	0.14 a	+[c]183 a	+40.6 c
75% Turfgrass	35.4	0.20 b	0.22	162 a	15.1 ab	0.18 a	+183 a	+20.7 b
90% Turfgrass	19.0	0.27 c	0.22	165 a	16.6 b	0.39 a	+168 a	+2.52 a

[a]Values within the same test parameter with the same letter are not significantly different at $P < 0.05$ using Tukey's HSD test in the quarter seasonal or annual budget.
[b]Soils were sampled from 0 to 15 cm depth.
[c]+ means nutrient surplus, - means nutrient deficit.
[d]numbers in this column are net P balances without soil P storage.

Table 3-3. Estimated volume of irrigation, rainfall, and leachate (quarterly or annually) for lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10).

Treatment	Irrigation	Rainfall	Leachate
	million L ha^{-1}		
Quarter one (5 Mar. 2011 to 3 June 2011)			
60% Turfgrass	0.38 a[a]	0.80	1.06 a
75% Turfgrass	0.50 b	0.80	1.16 a
90% Turfgrass	0.74 c	0.80	1.80 a
Quarter two (4 June 2011 to 2 Sept. 2011)			
60% Turfgrass	0.47 a	0.65	0.44 a
75% Turfgrass	0.56 ab	0.65	0.68 ab
90% Turfgrass	0.70 b	0.65	1.58 b
Quarter three (3 Sept. 2011 to 2 Dec. 2011)			
60% Turfgrass	0.23 a	0.77	0.41 a
75% Turfgrass	0.30 a	0.77	0.81 ab
90% Turfgrass	0.40 b	0.77	1.34 b
Quarter four (3 Dec. 2011 to 9 Mar. 2012)			
60% Turfgrass	0.27 a	0.13	0.00 a
75% Turfgrass	0.35 b	0.13	0.00 a
90% Turfgrass	0.53 c	0.13	0.43 b
Annual volume (5 Mar. 2011 to 9 Mar. 2012)			
60% Turfgrass	1.35 a	2.28	1.91 a
75% Turfgrass	1.71 b	2.28	2.64 a
90% Turfgrass	2.37 c	2.28	5.14 a

[a]Values within the same test parameter with the same letter are not significantly different at $P < 0.05$ using Tukey's HSD test in the same quarter or annual budget.

Table 3-4. Estimated temporal changes in volume of irrigation and leachate for lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10).

Quarter	Irrigation	Leachate
	million L ha^{-1}	
60% turfgrass treatment		
Q1[a]	0.38 b[b]	1.06 c
Q2	0.47 b	0.44 b
Q3	0.23 a	0.41 b
Q4	0.27 a	0.00 a
75% turfgrass treatment		
Q1	0.50 b	1.16 b
Q2	0.56 b	0.68 b
Q3	0.30 a	0.81 b
Q4	0.35 a	0.00 a
90% turfgrass treatment		
Q1	0.74 c	1.80 b
Q2	0.70 c	1.58 b
Q3	0.40 a	1.34 b
Q4	0.53 b	0.43 a

[a]Q1: 5 Mar. 2011 to 3 June 2011; Q2: 4 June 2011 to 2 Sept. 2011; Q3: 3 Sept. 2011 to 2 Dec. 2011; 3 Dec. 2011 to 9 Mar. 2012.
[b]Values within the same test parameter with the same letter are not significantly different at $P < 0.05$ using Tukey's HSD test in the same treatment

Table 3-5. Estimated temporal changes in N and P budgets for lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10).

Quarter	Irrigation N	Plant N removal	Leached N	Net N balance	Irrigation P	Mehlich 3 P[d]	Plant P removal	Leached P	Net P balance	Net P balance[e]
	kg ha^{-1}									
60% turfgrass treatment										
Q1[a]	0.58 a[b]	6.35ab	2.83 b	+[c]180b	0.04 a	117 a	1.14 a	0.08 c	+142ab	+24.9 c
Q2	1.03 b	24.2 c	0.81 ab	-22.8 a	0.06 b	139 ab	6.46 c	0.03 b	+132 a	-6.35 a
Q3	0.38 a	9.17 b	0.91 ab	+178 b	0.03 a	168 b	3.07 b	0.03 b	+191 b	+23.0 c
Q4	0.39 a	3.75 a	0.00 a	-3.19 a	0.03 a	148 ab	0.93 a	0.00 a	+147ab	-0.87 b
75% turfgrass treatment										
Q1	0.82 b	9.07 b	1.70 b	+123 c	0.05 a	120 a	1.80 a	0.10 a	+135 a	+15.9 c
Q2	1.24 c	27.1 c	1.23 b	-26.0 a	0.07 b	155 b	7.80 c	0.03 a	+147 a	-7.64 a
Q3	0.45 a	12.6 b	1.52 b	+119 c	0.04 a	197 c	4.11 b	0.05 a	+210 c	+13.7 c
Q4	0.51 a	4.00 a	0.00 a	-3.33 b	0.04 a	177 bc	1.21 a	0.00 a	+176 b	-1.13 b
90% turfgrass treatment										
Q1	1.17 c	7.00 b	4.06 b	+68.1 c	0.07 b	135 a	2.09 b	0.13 b	+142 a	+7.37 c
Q2	1.49 d	29.0 c	4.85 b	-31.3 a	0.08 c	167 b	9.15 d	0.12 b	+158ab	-9.08 a
Q3	0.64 a	8.41 b	3.51 b	66.3 c	0.05 a	181 b	3.93 c	0.11 ab	+187 b	+5.59 c
Q4	0.77 b	2.99 a	0.84 a	-2.88 b	0.07 b	178 b	1.40 a	0.04 a	+176 b	-1.34 b

[a]Q1: 5 Mar. 2011 to 3 June 2011; Q2: 4 June 2011 to 2 Sept. 2011; Q3: 3 Sept. 2011 to 2 Dec. 2011; 3 Dec. 2011 to 9 Mar. 2012.

[b]Values within the same test parameter with the same letter are not significantly different at $P < 0.05$ using Tukey's HSD test in the same treatment.

[c]+ means nutrient surplus, - means nutrient deficit.

[d]Soils were sampled from 0 to 15 cm depth.

[e]numbers in this column are net P balances without soil P storage.

Table 3-6. Mean and standard deviation for Mehlich 3 P, Fe, and Al concentrations of soils sampled on 7 Mar. 2012 under turfgrass and ornamental plant areas, respectively in lysimeters containing three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10).

Parameter	Turfgrass	Ornamental
	mg kg^{-1}	
Mehlich 3 P	59.2 ± 9.70	40.4 ± 21.9
Mehlich 3 Fe	61.8 ± 9.72	43.7 ± 12.3
Mehlich 3 Al	250 ± 33.7	261 ± 51.7

Table 3-7. Estimated annual percentages of nutrient outputs (N or P) of total applied nutrient inputs (N or P) for lysimeter plots planted with three mixed landscape treatments (turfgrass/woody ornamental proportions (%) of 60/40, 75/25 and 90/10) from 5 Mar. 2011 to 9 Mar. 2012.

Treatment	Leached N	Plant N removal	Leached P	Plant P removal
	%[b]			
60% Turfgrass	1.20 a[a]	11.4 a	0.26 a	22.1 a
75% Turfgrass	1.65 a	19.6 b	0.50 a	41.6 b
90% Turfgrass	8.24 b	29.5 c	2.02 b	85.1 c

[a]Values within the same test parameter with the same letter are not significantly different at $P < 0.05$ using Tukey's HSD test.
[b]Percentage of nutrient outputs were determined by dividing the weight of nutrients (N or P) in each output pathway by the total applied nutrients (N or P) weight. Soil stored available P was not considered as applied P inputs.

CHAPTER 4
CONCLUSIONS

The results of our study indicated that mixed landscapes containing higher proportion of turfgrass produced significantly higher leachate volumes, leachate nutrient concentrations (total Kjehldahl nitrogen, NH_4–N, nitrate N (NO_3 + NO_2–N), dissolved reactive phosphorus), and subsequently leached higher nutrient loads than landscapes containing a higher proportion of woody ornamental plants. We believe this was mainly due to the extensive root systems of the established woody ornamental plants in our lysimeters, which made ornamental plants very effective at absorbing water and dissolved nutrients from soils.

Using a nutrient budget method, we found out that fertilizer was the primary nutrient input and that plant nutrient removal was the main nutrient output in our study. Besides P inputs from fertilizer, soil P was also a substantial source for plant uptake. As a result, P fertilization of urban green areas should follow soil test recommendations to prevent adding excess nutrients; while N fertilization should follow published N rate recommendations. Also, fertilization of turfgrass and ornamental plants should follow different regimes that consider the nutritional needs of each plant type and the nutrient status of soils where turfgrass or ornamentals are planted. The nutrient (N and P) net balances in our study increased with the increasing of woody ornamental composition (60% turfgrass> 75% turfgrass> 90% turfgrass), which was mainly because the woody ornamentals were over-fertilized (when compared with UF-IFAS recommendations for woody ornamentals growing in the landscape). Although woody ornamentals received far more fertilizer than recommended, the treatment containing a higher proportion of woody ornamentals still leached significantly less nutrients (on a mass basis) than

treatments containing a higher proportion of turfgrass, suggesting that nutrient use efficiency was higher in landscapes with higher proportion of woody ornamental plants. We believe that the nutrient loss potential from mixed landscape containing higher proportion of woody ornamentals would be even lower if the fertilization rates had been more in line with recommended rates for landscape-grown plants. Also, the nutrient surplus condition in mixed urban landscape in our study can be ameliorated by reducing fertilizer inputs and splitting fertilizer application to apply lower rates at higher frequencies.

Although former studies indicated that landscape planted with establishing woody ornamental plants had significantly higher risk of leaching nutrient losses than landscapes covered with turfgrass alone, the high nutrient leaching potential of establishing woody ornamental plants can be controlled effectively by reducing fertilization and irrigation or matching the application of fertilizer and irrigation to the active plant uptake time. Moreover, our research showed that the inclusion of woody ornamental plants will result in a more environmental-friendly landscape in the long term, once the root systems of woody ornamentals expand fully. Therefore, we recommended that priority should be given to landscape types that contain higher proportion of woody ornamental plants when designing and managing urban green areas to reduce nutrients leaching losses, increase nutrient use efficiency, and subsequently control non-point source pollution from urban landscapes. This recommendation is consistent with urban best management practices promoted by the UF-IFAS Florida Friendly Landscaping™ (FFL) Program. One of the core principles of the FFL program is “right plant, right place”,

which often results in a reduction of turfgrass coverage and promotes the use of ornamental plants.

However, our study evaluated landscapes containing only one turfgrass species (St. Augustinegrass) and two woody ornamental plant species (magnolia and viburnum). Other popular warm-season turfgrass species (e.g., zoysigrass, bahiagrass, etc.) and ornamental plants (e.g., other woody species, herbaceous species) that are typical in urban green areas were not considered. Also, our study was conducted in Florida where sandy soil texture and intense precipitation events are common. As a result, leaching was the dominant pathway of nutrient loss to the environment in our study. However, other pathways (e.g., runoff, denitrification) that are of less concern in Florida may be a serious problem in other areas (e.g., areas with finer-soil texture, cooler climates). Future research should investigate nutrient losses from urban landscapes under various soil and climate conditions to determine the most regionally appropriate environmental-friendly combination of turfgrass and ornamental plants for urban pervious areas.

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BIOGRAPHICAL SKETCH

Zhixuan Qin grew up in a normal family in Changsha city in China. She started her education in one of the best elementary schools in Changsha city. Ever since high school, she had great interest in nature and was always curious about anything related to natural science. As a result, she decided to study ecology at South China Agricultural University (SCAU). During her undergraduate years, she received systematic education on subjects relevant to different components of the ecosystem including soil science, meteorology, animal science, microbiology, etc. She was fascinated with the interaction of the various components of the global ecosystem during her four-year learning process. While attending SCAU, she was selected as a research assistant by an ecology professor (due to her impressive academic record) to participate on a national project on bioremediation of polychlorinated biphenyls in soils. She gained precious experience in lab analysis and academic research during that process, which stimulated her interests to continue with graduate study. She was also selected as one of 28 students from College of Science and Agriculture to receive one-month study in agriculture, horticulture, ecology, and food science at the University of Hawaii in 2009. The study experience in America greatly broadened her eyes and helped her make the decision to have graduate study in United States.

She joined the interdisciplinary ecology graduate group at University of Florida with a concentration of Soil and Water science in 2010. This University of Florida offered her great learning and research experience to explore environmental issues and problems in mainly agricultural and urban areas. She was very happy and proud of her work at University of Florida, especially the fact that her work may potentially influence people to make use of the soil and water resources in an environmental-friendly way.

She will continue her graduate studies in a PhD program at the University of Delaware starting in fall 2012. Her goal is to keep on working on urban environmental issues and to apply what she learned to the field of water science and management.