Management impacts on nitrous oxide flux and water loss from urban ecosystems

Dissertation

Submitted in partial satisfaction of the requirements for the degree of

DOCTOR OF PHILOSOPHY

in Earth System Science

by

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2010
To my family,
who made this possible
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Abstract of the Dissertation

Management impacts on nitrous oxide flux and water loss from urban ecosystems

by Neeta S. Bijoor

Doctor of Philosophy in Earth System Science

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Urban ecosystems are heavily fertilized and irrigated, which causes emissions of nitrous oxide (N₂O), a powerful greenhouse gas, and consumption of water resources. Thus, it is important to (1) quantify the magnitude and drivers of N₂O losses, and their potential responses to climate change, (2) understand the amount and fate of irrigation water, including evapotranspiration (ET), (3) distinguish transpiration losses from evaporation, and (4) understand the water sources of urban plants, which may utilize irrigation water or groundwater for transpiration. To reduce these uncertainties for urban ecosystems in southern California, I made a variety of measurements including flux chamber measurements of N₂O and ET in experimental lawns, as well as the isotopic composition of plant, soil, water and atmospheric vapor, and soil moisture, irrigation and runoff in lawns, urban forests, and an urban wetland. This research has provided experimental
evidence that N₂O fluxes can serve as a positive feedback to global warming in lawns, due to a positive relationship with temperature. In addition, warming may exacerbate weed invasions, which may require more intensive management to manage species composition. I found that the irrigation efficiency (ET/applied irrigation) of urban lawns in this region may be very low (16-43%). The fate of more than half of the applied irrigation water at my study sites was drained below the rooting zone. Subsequently, the “reference ET” method of calculating ET of lawns from meteorological measurements likely over-estimates actual ET, and may lead to over-irrigation. I also found that despite frequent irrigation, urban trees tap into groundwater at some sites. However, mature urban trees may also develop very shallow roots, as I detected uptake of soil water above 30 cm depth. Finally, stable isotope techniques applied at an urban marsh provided support for primary water vapor loss by transpiration. A mass balance approach based on isotope measurements of standing water was more robust than a mass balance based on measurements of water vapor, due to a large influence of marine background vapor. The isotopic methods used in this research were powerful tools for understanding urban ecosystem nitrogen and water losses, especially when combined with flux measurements. My results show great potential for reducing N₂O emissions and water use in urban ecosystems in response to future climate change and water scarcity.
Chapter 1

Introduction

Most of the human population lives in urban ecosystems (UNFPA, 2007), and influences as well as is influenced by the urban environment. Urban land cover is projected to increase in extent by ~79% in the United States in the next 25 years (Alig et al., 2004). While it is known that biogeochemical cycles in urban ecosystems differ in many ways from natural ecosystems, there are still many uncertainties, including the transport of nitrogen and water to the atmosphere from urban environments (Kaye et al., 2006). Intensive fertilization and irrigation in urban ecosystems can cause significant gaseous emissions of nitrous oxide or N₂O (Kaye et al., 2004) and water loss to the atmosphere from evaporation and transpiration (Hurd, 2006). N₂O is an important greenhouse gas with a radiative forcing 296 times greater than that of carbon dioxide (Yung et al., 1976), and has been increasing in the troposphere at a rate of about 0.2% per year (Weiss, 1981, Khalil et al., 2002). An understanding of the magnitude and controls of N₂O emissions from urban ecosystems is therefore important for understanding urban contributions to future climate change. It is well-known that agricultural crops are large sources of N₂O at global, national, and statewide scales (Mosier et al., 1998, Bouwman et al., 2002), because fertilization increases the nitrogen substrate available for the microbial processes of nitrification and denitrification, which are responsible for N₂O flux. Urban soils are also heavily fertilized - 10% of all fertilizer nitrogen used in the USA is applied to urban
landscapes (USEPA, 1999, FAO, 2002 as cited in Qian et al., 2003); however, there have been surprisingly few direct measurements of N$_2$O emissions from urban soils (Maggiotto et al., 2000, Kaye et al., 2004, Bremer, 2006, Hall et al., 2008), such that the contributions of urban soils to local and regional greenhouse gas budgets are largely unconstrained.

The correlation between soil moisture and N$_2$O production (Brumme, 1995, Skiba et al., 1998, Flechard et al., 2007, Bijoor et al., 2008) highlights the role of water as a principal limiting resource for microbial activity, and suggests the importance of understanding the controls on evapotranspiration (ET), an important component of ecosystem water balance. Water is primarily lost from ecosystems by the process of ET, the sum of surface evaporation and the movement of water as vapor through plant stomates, or transpiration. Understanding losses of water from urban ecosystems has also become of increasing concern from a water conservation perspective, as competition for water resources is intensifying in semi-arid areas such as southern California (Reisner, 1993). Outdoor irrigation greatly influences urban water use, as 59 to 67% of total residential water demand is utilized for landscape irrigation in arid and semi-arid climates (Mayer, 2000). Uncertainties about urban ecosystem water use include an understanding of the fate of applied irrigation water, and accounting for all of the components of the water budget (Figure 1a). In addition, the sources of water to urban trees is highly uncertain, as irrigated trees may transpire applied irrigation water or tap into other sources such as groundwater (Figure 1b). Thus, knowledge about controls on water loss from urban ecosystems will be useful for the management of water resources.
Figure 1 – (a) Irrigation and precipitation may be evaporated, transpired, runoff, infiltrated into root zone, or drained below the root zone. (b) Irrigation, precipitation, groundwater, soil water, and/or runoff may serve as sources of water to trees. Arrow sizes are not proportional to magnitude of flows.

The difficulty of distinguishing transpiration from evaporation in water fluxes has led to frequent measurement of combined ET. The amount of irrigation water used by plants relative to the amount of water evaporated by soil is poorly understood, particularly in urban ecosystems. Stable isotopes have great potential to partition ET into its transpiration and evaporation components, because plant and soil water fluxes have distinct $\delta^{2}H$ and $\delta^{18}O$ signatures (Yakir et al., 1996, Wang et al., 2000, Yepez et al., 2005). The isotopic composition of transpiration at isotopic steady state (ISS) is equivalent that of the water source in the soil (Flanagan et al., 1991), and may differ substantially from the isotopic value of soil evaporation, which is strongly depleted in heavy isotopes (Craig et al., 1965, Gat, 1996).
To advance our understanding of these uncertainties, I undertook this work to understand N₂O losses, water budgets, water sources, and relationships among these variables. I studied nitrogen cycling and water budgets in turfgrass, the largest component of urban landscapes (Gleick, 2003) and the largest irrigated crop in the U.S. (Milesi et al., 2005). I then developed and applied stable isotope methodology in an urban marsh study site to separate water budget components, including separate quantification of evaporation and transpiration. In my final study, I investigated the sources of water from urban trees throughout the Los Angeles basin. I sought to answer the following questions related to my goal of understanding management impacts on N₂O and H₂O fluxes from urban ecosystems:

- **Chapter 2** - How does fertilization, temperature, and soil moisture impact turfgrass N₂O flux?
- **Chapter 3** - How does management intensity (irrigation technology and species composition) influence the water budget and irrigation efficiency (ET/applied irrigation)?
- **Chapter 4** - Can stable isotope mass balance techniques be used to infer how much irrigation water is lost via evaporation?
- **Chapter 5** - What are the sources of water to urban trees?

### 1.1. Specific objectives and hypotheses

I used a manipulative experiment in Chapter 2 to study the influence of temperature and fertilizer application on N₂O emissions and community composition in turfgrass (Bijoor et al., 2008). I used infrared lamps to increase surface temperature by 3.5±1.3°C. Control
and heated plots were split into high and low fertilizer treatments. I also measured the C/N (carbon/nitrogen ratio) of plants and nitrogen isotope ratio ($\delta^{15}$N) of soils, which can be used to infer gaseous losses of N. Carbon isotope ratio ($\delta^{13}$C) was used to delineate between proportion of C$_3$ fescue vs. C$_4$ crabgrass weeds in the plots, as these photosynthetic pathways have distinct isotopic signatures. As increased temperature is expected to increase metabolic activity of microbes, I hypothesized that warming as well as fertilization and soil moisture would increase N$_2$O losses in this ecosystem (Brumme, 1995, Skiba et al., 1998, Flechard et al., 2007). I expected plant C/N to decrease in response to fertilizer addition, but remain relatively constant in response to warming due to the large amounts of nitrogen applied to this system. I did not expect a change in plant community composition. My measurements were conducted to understand the magnitude and controls of N$_2$O emissions and nitrogen cycling in turfgrass, and how they might alter with expected climate change.

In Chapter 3, I continued my work in turfgrass, this time to understand the fate of irrigation water in landscapes with “smart” irrigation controllers, which are expected to reduce water use (Devitt et al., 2008). In this study, I took direct measurements of landscape water use and plant function to understand their role in water budget. I compared irrigation rates, soil moisture, leaf-level stomatal conductance, and complete water budgets of three landscapes managed with varying intensity. The “Typical” landscape was expected to have the highest irrigation rate, soil moisture, ET, stomatal conductance, and the lowest irrigation efficiency, as it utilized a conventional irrigation system on a timer, and had a cool-season, fescue lawn. In contrast, the Low-impact
landscape was expected to have the lowest irrigation rate, soil moisture, and highest irrigation efficiency, as this landscape utilized a weather station based irrigation system, and a native sedge lawn. The warm-season grass in the Retrofitted landscape was expected have the lowest ET, stomatal conductance, and the highest growth rate (Huang et al., 2000, Robins, 2010). This system utilized a soil moisture sensor-based irrigation system. In general, I sought to better understand how management influences plant function and water budget components with direct measurements, in order to inform urban ecology, hydrology, and landscape management.

I developed and applied two stable isotope approaches to quantify water budget components, and tested them at an eddy-covariance site in Chapter 4. Stable isotope techniques have an advantage over most methods because they have the ability to distinguish evaporation from transpiration (Zhang et al., 2010). The ET partitioning approach was applied over a managed urban wetland, which was an ideal test system because it was fairly homogenous and had possible large water vapor sources from evaporating standing water as well as transpiring plants. In addition, direct measurements of ET were available. The stable isotope approaches were based on the mass balances of standing water and water vapor at the marsh. I investigated leaf water isotopic composition to determine if plants were transpiring in steady state. This information was needed to determine the isotopic composition of transpiration. I hypothesized that plant transpiration would be primarily in isotopic steady-state, which has been shown for short canopies (Yakir & Wang 1996, Welp et al., 2008). I expected that transpiration would dominate ET, as was previously reported for this site (Goulden et al., 2007).
Consequently, I expected that a small percentage of total marsh water loss would be due to evaporation, and that the largest water losses would be transpiration and subsurface drainage. In general, I sought to assess the extent to which isotope mass balance studies can inform wetland ecology and ecohydrology.

I expanded my work to urban ecosystems throughout the Los Angeles Basin in Chapter 5. In this study, I used stable isotopes to understand the sources of water to urban trees. I measured the isotopic composition of possible tree water sources at several horticultural settings in the Los Angeles basin, including irrigation water, rain water, and groundwater. I also measured both the temporal and spatial variability in the isotopic composition of soil water and stem water. I hypothesized that the isotopic composition of irrigation would change very little over time, and that soil water would consist of evaporated irrigation water. I also hypothesized that trees would rely on soil water rather than groundwater due to frequent irrigation which would likely result in shallow rooting depths. Finally, I expected that the isotopic composition of tree stem water would not change over time, except in response to large precipitation events. The results of this study can inform a general understanding of the ecohydrology of irrigated, semi-arid cities as well as urban forest planning and management.
Chapter 2

The effects of temperature and fertilization on nitrogen cycling and community composition of an urban lawn

As appears in:

2.1. Abstract

We examined the influence of temperature and management practices on the nitrogen (N) cycling of turfgrass, the largest irrigated crop in the U.S. We measured nitrous oxide (N\textsubscript{2}O) fluxes and plant and soil N content and isotopic composition with a manipulative experiment of temperature and fertilizer application. Infrared lamps were used to increase surface temperature by 3.5 ± 1.3°C on average and control and heated plots were split into high and low fertilizer treatments. N\textsubscript{2}O fluxes increased following fertilizer application and also were directly related to soil moisture. There was a positive effect of warming on N\textsubscript{2}O fluxes. Soils in the heated plots were enriched in nitrogen isotope ratio (δ\textsubscript{15}N) relative to control plots, consistent with greater gaseous losses of N. For all treatments, C\textsubscript{4} plant C/N ratio was negatively correlated with plant δ\textsubscript{15}N, suggesting that
low leaf N was associated with use of isotopically depleted N sources such as mineralized organic matter. A significant and unexpected result was a large, rapid increase in the proportion of C₄ plants in the heated plots relative to control plots, as measured by the carbon isotope ratio ($\delta^{13}C$) of total harvested aboveground biomass. C₄ plant biomass was dominated by crabgrass, a common weed in C₃ fescue lawns. Our results suggest that an increase in temperature caused by climate change as well as the urban heat island effect may result in increases in N₂O emissions from fertilized urban lawns. In addition, warming may exacerbate weed invasions, which may require more intensive management, e.g. herbicide application, to manage species composition.

2.2. Introduction

Turfgrass ecosystems are expanding rapidly in conjunction with urbanization, which is expected to increase 79% in the United States in the next 25 years (Alig et al., 2004). As the largest irrigated crop in the U.S., turfgrass currently covers 1.9% of the national surface area (Milesi et al., 2005). Turfgrass land cover can sequester carbon (C) (Qian et al., 2002), but can also cause significant emissions of nitrous oxide (N₂O) (Kaye et al., 2005), an important greenhouse gas that has a radiative forcing 296 times greater than that of carbon dioxide (CO₂) (IPCC, 2001), and that has been increasing in the troposphere at a rate of about 0.2% per year (Weiss, 1981, Khalil et al., 2002). It is therefore important to understand nitrogen (N) cycling and N₂O emissions from turfgrass, and their potential responses to climate change.
It is difficult to predict how warming caused by climate change and the urban heat island effect (Arnfield, 2003) may influence N cycling of turfgrass. It is also difficult to predict how plant N concentrations will change in response to warming; field experiments have shown that foliar C/N changes can be dynamic or species-specific (Read et al., 1996, Klein et al., 2007). Experimental warming manipulations in various ecosystems have shown that warming may significantly alter soil N budgets through increased N mineralization (Shaver et al., 2000, Rustad et al., 2001). However, the effects of elevated temperature on nitrification and denitrification and subsequent N₂O emissions are still unclear due to the small number of field studies (Barnard et al., 2005). The impact of warming on N₂O fluxes is complicated by two possible effects – reduced soil moisture, generally a negative effect, and increased soil temperature, a positive effect (Brumme, 1995; Skiba et al., 1998; Flechard et al., 2007). Studies of fertilized soils are particularly important, as agricultural crops are known to be large sources of N₂O at global, national, and statewide scales (Mosier et al., 1998, Bouwman et al., 2002, Franco, 2002). There have been surprisingly few direct measurements of N₂O emissions from turfgrass (Maggiotto et al., 2000, Kaye et al., 2004, Bremer, 2006), such that the contributions of managed lawns to local and regional greenhouse gas budgets are largely unconstrained.

We initiated an experimental warming and fertilization manipulation in a fescue dominated lawn to understand how temperature, fertilizer, and their combination influence N cycling. In addition to measuring soil N₂O fluxes, we examined foliar and soil N content and nitrogen isotope ratios (δ¹⁵N) in each treatment in order to determine effects on plant N availability. δ¹⁵N is a useful indicator of plant N sources and
ecosystem N losses, as N derived from mineralized organic matter may undergo microbial fractionation (Shearer et al. 1974; Mariotti 1980; Nadelhofer and Fry 1988), and is likely to be lighter than fertilizer-derived N, which is often isotopically enriched due to gaseous N loss (Högberg 1990; Högberg 1997; Robinson 2001). Our study system was a commonly occurring mixture of C3 fescue and C4 crabgrass, therefore, we measured the carbon isotope ratio ($\delta^{13}C$) of each species separately and in total harvested aboveground biomass to quantify the relative abundance of each species in each treatment. The $\delta^{13}C$ of C4 biomass varies from -11 to -15 ‰, while the $\delta^{13}C$ of C3 biomass varies from -20 to -35 ‰ (Dawson et al., 2002). The isotopic composition of each functional type is isotopically distinct, thus, measurements of $\delta^{13}C$ may be used to determine the proportion of C3 vs. C4 plants in each treatment.

As increased temperature is expected to increase metabolic activity of microbes, we hypothesized that increased temperature as well as fertilization would increase N2O fluxes in this ecosystem, with a corresponding enrichment of $\delta^{15}N$ in the soil. We also hypothesized that plant C/N would decrease in response to fertilizer addition and then remain relatively constant in response to warming due to the large amounts of fertilizer N applied to this system. While in general, plants utilizing the C4 photosynthetic pathway have a competitive advantage over C3 plants in warmer environments based on the relationship between the quantum yield of photosynthesis and temperature (Ehleringer and Bjorkman, 1977), we did not expect the proportion of C3 and C4 biomass to change greatly in our 15 month measurement period. Changes in the proportion of C3 and C4 plants may be very important for turfgrass management: C4 plants are often weeds in C3
lawns, requiring the application of herbicides to turfgrass in addition to fertilizer. Hence, the impacts of warming and fertilization on N₂O fluxes, ecosystem N cycling, and species composition are highly relevant for turfgrass ecosystem management as well as global change.

2.3. Materials and Methods

2.3.1. Study site

This study was conducted at the University of California, Irvine, Arboretum (33.7° N 117.7° W, 30 m a.s.l.) on a turfgrass lawn dominated by tall fescue (*Schedonorus phoenix* (SCOP.) HOLUB), a cool season, C₃ species, and crabgrass (*Digitaria HALLER*), a warm season, C₄ species. The site has a Mediterranean climate, with a mean annual air temperature of 18.6°C and 352 mm of precipitation, primarily falling between November and April. Tall fescue is a widespread turf species in the U.S., adapted to cool and humid climates (USDA-NRCS, 2007). Crabgrass is an annual weed that commonly invades domesticated lawn grasses. Bermudagrass (*Cynodon dactylon* L.), a warm season, C₄ species, was initially present at the site at low abundance. The experimental site was fenced to exclude grazing by wildlife, primarily rabbits. The soil type was alkaline aloclay (USDA-NRCS, 2007). Prior to the experiment, this site was managed as turfgrass for over sixteen years (L. Lyons, pers. comm. 2008).

2.3.2. Experimental Design

Six plots of 1.5 x 2 m were established on June 10, 2005. Pairs of plots were blocked by slope position at three levels on a slight slope. For each pair, plots were randomly
assigned to control or heated treatments. Each plot was split into two subplots that were randomly assigned a low or high fertilizer treatment. In March 2006, these subplots were separated with 10 cm deep plastic trenching material inserted into the soil. Measurements were concentrated in the central 2,500 cm² of the subplots, while the surrounding area served as a buffer zone. This design resulted in a 2 x 2 factorial experiment of fertilizer and temperature with 3 replicates (total of 12 plots). In this paper, the treatments will be referred to as control (C), high temperature x low fertilizer (T), control temperature x high fertilizer (CN), and high temperature x high fertilizer (TN). The plots were watered with domestic water approximately three times a week during summer (May to September) and twice a week during the winter. Starting in July 2006, all plots received exactly equal amounts of water based on recommendations by Hartin et al. (2001). The heated subplots were subjected to an average increase in mean daily surface temperature of 3.5±1.3°C (average ± SD) by elevating ceramic infrared heaters (250W, Exo Terra, Rolf C. Hagen Inc., Montreal, Canada) 1 m over the ground. Plots were heated from July 14, 2005 to December 1, 2006. The applied fertilizer was a common commercial formula of 29:3:4 NPK that contained 28% CON₂H₄, 1% NH₄, 3% P₂O₅, 4% K₂O, and 1% Fe (Vigoro Ultra Turf, Spectrum Group, St. Louis, MO, USA). The δ¹⁵N of fertilizer was -0.64 ± 0.09 ‰. Low fertilizer plots received 76.4 kg N ha⁻¹yr⁻¹ in two applications in 2006 (62.3 kg N ha⁻¹ on April 2, and 14.0 kg N ha⁻¹ on July 20) and high fertilizer plots received 118.5 kg N ha⁻¹yr⁻¹ in four applications in 2006 (62.3 kg N ha⁻¹ on April 2, 14.0 kg N ha⁻¹ on May 21, 28.1 kg N ha⁻¹ on July 20, and 14.0 kg N ha⁻¹ August 31). The treatments were based on the recommended fertilizer application rates for these varieties, which vary from approximately 50 to more than 200 kg N ha⁻¹ yr⁻¹ divided into 2-6
applications during March to November (Reynolds et al., 2004). These are also plausible fertilization scenarios for householders based on Osmond & Hardy (2004), who found that five North Carolina communities apply 24 to 151 kg N ha\(^{-1}\) to turfgrass and had on average between 1.5 and 3 applications per year.

2.3.3. Measurements

Soil surface temperature (\(T_s\)) and volumetric soil water content (\(\theta\)) were measured in the center of each plot. Surface temperature was measured at the mineral soil surface with copper-constantan thermocouples and \(\theta\) was measured at 5 cm soil depth with water content reflectometers (CS616, Campbell Scientific, Inc., Logan, UT, USA). Air temperature and relative humidity were continuously recorded at 1 m above ground in one location at the site starting March 22, 2006 (CS215 shielded by 41303-5a, Campbell Scientific). Prior to this, meteorological variables were obtained from the California Irrigation Management Information System (www.cimis.water.ca.gov, Irvine station #75). All environmental data were recorded every 30 min (AM25 and AM16/32 multiplexer, CR10x logger, Campbell Scientific).

The rate of N\(_2\)O efflux from the soil surface was measured using a static, poly-vinyl chloride (PVC) chamber lid (height of 15 cm and inner diameter of 26 cm) containing a septum port. The chamber was placed over the soil surface and the lower rim was surrounded by water-filled plastic tubes to prevent diffusion of ambient air into the chamber. Gas samples were taken using a syringe at four timed intervals over a 15 or 21-minute period, and were injected into air-tight, pre-evacuated 12 mL vials. The samples
were shipped to the University of Kansas where they were analyzed for N₂O on a Varian CP3800 gas-chromatograph fitted with a $^{63}$Ni electron capture detector, operated at high temperature (300-400°C) using N₂ as the carrier gas. Rates of N₂O-N loss were calculated as the rate of N₂O accumulation over time in the chamber. Temperatures inside the N₂O chambers were assumed to be equivalent with ambient air temperatures measured at the site.

Aboveground biomass production was clipped approximately monthly to a height of 4 cm at the same time that the area outside the plots was mowed to the same height (clippings removed). The biomass was oven-dried at 70°C for at least 48 hours, and weighed. Subsets of well-mixed bulk biomass, C₃ only, and C₄ only biomass material were removed from each harvest for C, N, and stable isotope analysis. Specific leaf area (SLA) was estimated on August 10, September 11, and October 12, 2006 by determining leaf area on a subset of harvested fresh leaves (ImageJ software, U.S. National Institute of Health, http://rsb.info.nih.gov/ij/) and dividing leaf area by dry weight. Another subset of samples were ground to a fine powder and analyzed for % C, % N, $\delta^{13}$C and $\delta^{15}$N with an elemental analyzer (Carlo Erba NA 1500 NC, Milan, Italy) coupled to an isotope ratio mass spectrometer (Thermofinnigan Delta Plus, San Jose, CA, USA) at the University of California, Irvine stable isotope facility. Isotope ratios were referenced to the PDB standard for C and the atmospheric standard for N. The precision of these measurements was 0.14, 0.06, 0.18, and 1.00 (SD) for $\delta^{15}$N, $\delta^{13}$C, % N, and % C, respectively.
Soil samples were collected on July 17 in 2005 and June 6 and December 1 in 2006 at 0-5 cm and 5-10 cm depths. In 2005, pre-treatment samples were collected from the buffer zone of each plot before the plots were split into high and low fertilizer treatments (total of 6 samples). In June 2006, one sample was taken from the buffer zone of each subplot (total of 12 samples). Samples were collected from the plot buffer zone to minimize disturbance; measurements of the spatial variability of surface temperature in the plots indicated that the soil in the buffer zone was subjected to a similar increase in temperature as the main measurement area. Soil samples from 2005 were oven dried at 60°C, and samples from 2006 were freeze-dried. The different processing methods were related to other measurements not reported here and should not affect the determination of total soil C, N, and isotopic composition. After removal of roots and litter, sub-samples were ground to a fine powder and acid-treated for removal of inorganic C. Two grams of each sub-sample were treated with 0.5 M HCl and shaken overnight, centrifuged to remove the remaining acid, washed with 20 mL of water, centrifuged and decanted, and freeze-dried. The acid-treated soils were analyzed for C and N content and isotope ratios. The precision of these measurements was 0.20, 0.06, 0.23, and 1.64 (SD) for δ¹⁵N, δ¹³C, % N, and % C, respectively.

2.3.4. Statistical analysis

Data were analyzed using SAS software v. 9.1 (SAS Institute, Cary, NC). Split-plot, repeated measures ANOVAs were used to determine treatment effects, as well as analyses of covariance (ANCOVAs), when volumetric soil water content co-varied with the dependent variable. Paired t-tests were used to analyze pre- vs. post-treatment
differences, and ANOVAS were used to analyze differences among treatments for individual sampling periods. For all analyses, p-values <0.05 were considered significant.

2.4. Results

2.4.1. $\text{N}_2\text{O}$ fluxes

In all treatments, $\text{N}_2\text{O}$ fluxes were highly variable throughout the year (Figure 1a). Heated plots had higher fluxes than control plots following the first and second fertilization events on April 3, April 7, and May 22, 2006 (p=0.0183, p= 0.0360, and p<0.0001). High N plots had significantly higher fluxes than low N plots on May 22 and July 27, 2006 (p=0.0148 and p=0.0327). The highest fluxes were 2.0 mg N m$^{-2}$d$^{-1}$, and were measured in the heated (T and TN) subplots following fertilization in April 2006. The temporal pattern of $\text{N}_2\text{O}$ fluxes appeared to follow the seasonal pattern in daily average $\theta$ in the growing season (Figure 1b). Before June, when average daily $\theta$ was 39.4 ± 0.42 %, $\text{N}_2\text{O}$ fluxes increased immediately following fertilization events, and then decreased (Figure 1a). Fluxes remained low despite two additional fertilizer applications after June, coincident with a decrease in soil moisture. Although the lawn was watered intensively during summer, soil moisture declined due to unusually high temperatures and high evaporative demand (Czimczik et al., submitted).
Figure 1 - (a) N₂O fluxes in control (C), high temperature (T), high N (CN), and high N x high temperature (TN) treatments. Dotted lines indicate fertilization events. Warming treatment began July 14, 2005 (not shown). The asterisk (*) shows a significant treatment difference due to temperature, and circles (o) show treatment differences due to fertilization. Error bars show the standard error. (b) Average volumetric soil water content (%) in each treatment.
N$_2$O fluxes were positively correlated with soil moisture in all four treatments (Figure 2). An analysis of covariance (ANCOVA) with soil moisture as the covariate showed that there was no difference in the slope of this relationship among C, CN, and T treatments (p>0.05); however, TN had a greater slope than the other treatments. N$_2$O fluxes measured from T were higher than C and CN (p=0.0391 and 0.0046, respectively). N$_2$O fluxes measured from C and CN were not significantly different (p>0.05). For all treatments, there was a decline in N$_2$O fluxes over time (p<0.0001), indicating that fluxes...
decreased with the decline in soil moisture in summer, probably caused by an increase in soil temperature during this period (Czimczik et al., submitted). There were interactions of time with temperature on N$_2$O fluxes before and after the initiation of high fertilizer treatments (p=0.0003 and p=0.0002), showing that N$_2$O fluxes from warmed plots initially increased, but then decreased when soil moisture was low. The integrated flux over the measurement period was 75.2 mg N m$^{-2}$ for C and 107.7 mg N m$^{-2}$ for T. After initiation of high fertilizer treatments, the integrated flux was 16.8, 11.4, 32.6, and 16.3 mg N m$^{-2}$ from C, CN, T, and TN, respectively.

2.4.2. Plant isotopic and chemical composition

In C$_4$ plants, leaf C/N increased with time prior to fertilization of high N treatments (CN and TN), but then did not change afterwards (Figure 3a and Table 1; repeated measures ANOVA, p=0.0002 and p>0.05, respectively). The application of fertilizer in the high N treatments decreased leaf C/N of C$_4$ plants (Table 1 and Figure 3a; p=0.0028). On October 12, 2006, there was a marginally significant decrease in C/N of C$_4$ plants in the high N plots (p=0.0613). There was no effect of temperature on C/N in C$_4$ plants (Table 1; p>0.05), although on October 12, 2006, heated plots had lower C/N (p=0.0498). In C$_3$ plants, leaf C/N increased with time prior to fertilization of high N plots, and then decreased with time afterwards (Figure 3b and Table 1; repeated measures ANOVA, p<0.0001 and p=0.0021, respectively). At the end of the experiment, C$_3$ plants in heated (T and TN) plots had lower C/N than in the control (C and CN) plots (Figure 3b; p=0.0410). After establishment of CN and TN plots, C/N of C$_3$ plants were lower than those of C$_4$ plants (repeated measures ANOVA, p<0.0001). There was a negative
Figure 3 - C/N ratio of (a) C₄ aboveground biomass and (b) C₃ aboveground biomass. Dotted lines indicate fertilization events. Asterisks (*) show significant treatment differences due to temperature, and circles (o) show treatment differences due to fertilization. Treatments are abbreviated as in Figure 1. Error bars show the standard error.

Table 1 - F- and p-values for treatment effects on C₃ and C₄ foliar C/N for time periods before and after fertilization of high N plots (CN and TN). Significant effects (p < 0.05) are marked by asterisks (*).
correlation between C/N and $\delta^{15}$N of aboveground biomass for C$_4$ grasses in each treatment (Figure 4). This relationship was particularly strong in the C treatment, where leaf $\delta^{15}$N was also the most isotopically depleted, with a value as low as -2.9 ‰. An analysis of covariance (ANCOVA) with C/N as the covariate showed that there was no difference in the slope among treatments ($p>0.05$). Heated plots were more enriched in $\delta^{15}$N ($p<0.0001$) with no effect of fertilization ($p>0.05$).

![Figure 4](image)

**Figure 4** - The relationship between $\delta^{15}$N and C/N ratio of C$_4$ aboveground biomass in each treatment. Treatments are abbreviated as in Figure 1. Symbols represent different blocks. $P$ is <0.0001, 0.0446, 0.0334, 0.0013 for C, CN, T, TN respectively; $R^2$ were 0.66, 0.32, 0.18, 0.56 for C, CN, T, and TN treatments respectively.
For C₃ plants, C/N and δ¹⁵N were correlated only in the T treatment (not shown; p=0.0019). In the C treatment, there was a significant relationship between δ¹³C of C₃ plant biomass and soil moisture (Figure 5a; p=0.0021), and a marginally significant relationship for the CN treatment (Figure 5b; p=0.0507).

![Figure 5 - Average volumetric soil water content prior to harvest vs. δ¹³C of C₃ aboveground biomass in (a) control (C) and (b) high N (CN) treatments. Symbols distinguish different blocks.](image)

All treatments showed increases in C₄ aboveground biomass with time as estimated with δ¹³C of bulk harvests (repeated measures ANOVA on δ¹³C; p<0.0001). Fertilization and its interactions were non-significant for δ¹³C (p>0.05), such that fertilization treatments were combined to evaluate the effect of warming (Figure 6a). Prior to the warming treatment, the δ¹³C of bulk harvests from both control and treatment plots were similar (-25.6 ± 0.7 ‰ and -24.5 ± 1.2 ‰, respectively), indicating similar proportions of C₃ vs. C₄
species. In a period of approximately seven months, the aboveground biomass in heated plots became isotopically enriched by 4.9 ‰ on average (Figure 6a), indicating a greater proportion of C₄ plant material. Repeated measures ANOVAs showed no effect of warming on δ¹³C before January 31, 2006 (p>0.05), but an effect of warming on δ¹³C following January 31, 2006 (p=0.0275). The isotope ratios of each species measured at each harvest were utilized as end-members to calculate the proportion of C₃ versus C₄ biomass, and confirmed that changes in δ¹³C of the bulk harvest were dominated by changes in community composition, rather than changes in δ¹³C of C₃ plants (Figure 6b). C₄ biomass was 30 ± 16 % greater in the heated plots by October 2006 (Figure 6b).

**Figure 6** - (a) δ¹³C of bulk turfgrass aboveground biomass. (b) The percentage of total aboveground biomass comprised of C₄ plants as calculated from the isotopic composition of C₃ biomass, C₄ biomass, and the bulk harvest. The dashed line indicates the beginning of the warming treatment, and dotted lines indicate fertilization events. High and low N treatments are combined as they were not statistically different. Asterisks (*) show significant treatment differences due to temperature. Error bars show the standard error.
2.4.3. Soil chemical and isotopic composition

There was no change in N and $\delta^{13}$C of soil in response to fertilization, temperature, or time (Figure 7a,c, $p>0.05$). However, after the warming treatment was applied, soil $\delta^{15}$N in heated plots were enriched relative to control plots at the 0-5 cm depth (Figure 7b, $p=0.016$ in June 2006) and at the 5-10 cm depth (Figure 7b, $p=0.031$ in June 2006 and $p=0.007$ in Dec. 2006).

![Figure 7](image)

**Figure 7** – (a) N (kg m$^{-2}$), (b) $\delta^{15}$N, and (c) $\delta^{13}$C of soil measured at two depths at the beginning of the warming treatment (Jul 2005), following fertilization (Jun 2006), and at the end of the experiment (Dec 2006). Treatments are abbreviated as in Figure 1. Asterisks (*) show significant treatment differences due to temperature. Error bars show the standard error.

2.4.4. Plant canopy properties

Specific Leaf Area (SLA) averaged $181.9 \pm 11.1$ cm$^2$ g$^{-1}$ (mean ± SE) in C$_3$ plants and $217.8 \pm 5.6$ (mean ± SE) in C$_4$ plants. There were no significant effects of temperature, fertilizer, time, or their interactions on SLA for C$_3$ or C$_4$ plants (repeated measures
ANOVA, p>0.05). Leaf area index (LAI) changed significantly over time for C3 and C4 plants (p=0.0367 and p=0.0006 respectively), but was not affected by fertilization (p>0.05). C3 plants in heated plots had lower LAI than control plots on two sampling dates (Aug 10 and Sept 11, p=0.0088 and p=0.0422 respectively). LAI of C4 plants was unaffected by heating. Average total plot LAI in 2006 ranged from 0.60 ± 0.08 m² m⁻² (mean ± SE) in August-September to 0.80 ± 0.11 m² m⁻² (mean ± SE) in October.

2.5. Discussion

Measured N₂O fluxes were within the reported range for turfgrass (Maggiotto et al., 2000, Kaye et al., 2004, Bremer, 2006). As hypothesized, fertilizer application caused an increase in N₂O fluxes, although the effects were significant only on two sampling dates. Maggiotto et al. (2000) and Bremer et al. (2006) also reported increased N₂O emissions from fertilized turfgrass plots relative to control plots. The positive effect of warming on N₂O fluxes was consistent with our hypothesis. We are not aware of prior warming manipulations in turfgrass, and field warming studies in other ecosystems have shown conflicting effects on N₂O fluxes. Kamp et al. (1998), who applied a 3°C warming to wheat and fallow fields, did not find differences in cumulative emissions of N₂O, but did find that heated fallow plot emissions were 3 times higher than control plots during summer. Conversely, Hantschel et al. (1995) found lower N₂O fluxes in 3°C heated wheat fields during winter in Germany. Peterjohn et al. (1994) did not find an effect of a 5°C warming on N₂O fluxes in a deciduous hardwood forest. McHale et al. (1998) did not find a strong response of heating on N₂O fluxes in plots heated to 2.5, 5.0, or 7.5°C above ambient in a northern hardwood forest. In a companion study, we found increases in
ecosystem respiration in heated plots relative to control plots in winter when aboveground biomass production was low (Czimeczik et al., submitted). It is possible that heating increased heterotrophic respiration and N-mineralization. Higher rates of N-mineralization in heated plots may have increased available ammonium pools, causing the positive response of N₂O fluxes to heating. In summer, N₂O fluxes declined, perhaps because low soil moisture suppressed the hydrolysis of applied urea fertilizer to ammonium, the substrate for N₂O production via nitrification. The activity of urease, the enzyme responsible for urea hydrolysis, peaks near field capacity and declines with decreasing soil moisture (Vlek et al., 1983, Sahrawat, 1984). In addition, urea hydrolysis strongly depends on incorporation of urea into soil through diffusion of dissolved ammonia (Sadeghi et al., 1989). Thus, it is possible that fertilizer was unavailable for microbial processes during the period of low soil moisture in summer, resulting in low N₂O fluxes (Figure 1). A reduction in soil denitrification, which requires anoxic conditions associated with soil moisture, may also explain the decline of N₂O fluxes when soil moisture was low.

Leaf C/N of C₃ plants decreased in the period fertilization of high N plots, supporting our hypothesis (Figure 3b). However, C/N of C₄ plants did not change over time in response to fertilization, contrary to our hypothesis (Figure 3a), indicating plant N-limitation despite the additional application of fertilizer. The negative correlation between leaf C/N of C₄ grasses and leaf δ¹⁵N (Figure 4) suggests greater use of isotopically light forms of N when N was limiting. The labile soil organic matter that is being fractionated by mineralization (Shearer et al., 1974; Mariotti, 1980, Nadelhoffer et al., 1988) is likely to
be isotopically lighter than fertilizer N, which can undergo isotopic enrichment due to rapid gaseous loss through volatilization (Högberg, 1990, Högberg, 1997, Robinson, 2001). As the soils in this study were alkaline (pH ranging from 7-9), ammonia volatilization may have been an important pathway for gaseous N loss (Kirchmann et al., 1989). Thus, the correlation between C/N of C4 grasses and $\delta^{15}$N in all treatments suggests greater use of fertilizer-derived ammonium during wetter conditions earlier in the year, versus uptake of N mineralized from organic matter late in the year when applied fertilizer was not biologically available. Additional measurements of inorganic N forms and their isotopic composition would be required to validate this interpretation, as many processes may cause fractionation of plant and soil N (Evans, 2001, Robinson, 2001). Plants in the high N treatments have shown greater isotopic enrichment and weaker correlations between $\delta^{15}$N and C/N than the C treatment, possibly because of increased fertilizer uptake. The T treatment also showed a weaker trend and greater isotopic enrichment than the C treatment, possibly because of greater N$_2$O and other gaseous losses such as NH$_3$, NO, and N$_2$ (Figure 1). This is supported by the soil data, which shows enrichment of $\delta^{15}$N in heated plots (Figure 7b).

$\delta^{13}$C of leaves is determined by $c_i/c_a$, the ratio of CO$_2$ inside and outside of the leaf, which is in turn determined by the balance between photosynthesis and stomatal conductance (Farquhar et al., 1989). Previous studies have reported more positive $\delta^{13}$C values in response to reduced soil moisture in C$_3$ grasses, including tall fescue turfgrass (Johnson et al., 1991, Ebdon et al., 1998, Johnson et al., 1999). In the C and CN treatment, we found a direct correlation between $\delta^{13}$C and soil moisture in the C$_3$ plants
(Figure 5), which suggests stomatal closure in response to declining soil moisture. In the T and TN treatment, there was no correlation with variations in soil moisture. However, \( \delta^{13}C \) of C3 plants was significantly more enriched in the two high temperature treatments than the C and CN treatments (ANOVA, \( p= 0.0385 \)), suggesting that plants in the T and TN treatments were generally more water stressed, consistent with the lower soil moisture (Figure 1b).

One of the most significant results of this study was unexpected: the temperature treatments facilitated a rapid (~ 7 mo.) shift in C4 weed biomass relative to C3 plant biomass, as shown by the time series of \( \delta^{13}C \) of total biomass (Figure 6a). C4 biomass was 30 ± 16 % greater in the heated plots at the end of the experiment in October 2006 (Figure 6b). Immediately following the first spring fertilizer application, \( \delta^{13}C \) of total biomass in the control and high temperature plots had become more similar and were not significantly different, possibly because fertilization enabled the C3 plants to compete more successfully with the C4 species, which have higher N-use efficiency (Sage et al., 1987). A greater ability to take up isotopically heavy N early in the growing season may also have contributed to the competitive advantage of C4 species in all plots, and particularly in heated plots, and may explain why C4 C/N did not change over time. Since C3 and C4 plants can have different patterns of N uptake and allocation, changes in C3/C4 community composition could ultimately influence other aspects of N cycling such as \( N_2O \) fluxes, as both microbes and plants share the same soil N pool.
Plants utilizing the C$_4$ photosynthetic pathway have a competitive advantage over C$_3$ plants in warmer environments (Ehleringer et al., 1977). Because most weeds are C$_4$ plants (Holm, 1977), there is great concern about the possibility of more widespread weed invasions as a result of global climate change (Patterson, 1995, Dukes et al., 1999, Sage et al., 2003, Schmitz, 2006). However, field experiments often fail to confirm simple predictions based on physiological principles. For example, C$_4$ plants were predicted to be relatively unresponsive to elevated atmospheric CO$_2$, but this has been contradicted by experimental evidence (Dukes et al., 1999). As most field warming experiments have been conducted in high latitude ecosystems where C$_4$ plants are largely absent, there have been few of these studies on C$_3$ vs. C$_4$ plants (White et al., 2000, White et al., 2001, Wan et al., 2005, Luo, 2007), particularly in agricultural and highly managed ecosystems, e.g. (White et al., 2000, Ziska, 2000, White et al., 2001, Derner et al., 2003, Fuhrer, 2003, Ziska, 2003). Weed expansion under higher temperatures is particularly relevant because herbicide is commonly applied to fescue lawns to control the invasion of C$_4$ weeds such as crabgrass, often at higher application rates than in other types of agriculture (Templeton et al., 1998).

2.6. Conclusions

This study showed reduced N$_2$O fluxes and plant N limitation with declining soil moisture and increases in N$_2$O fluxes with warming. Negative correlations between C/N of C$_4$ plants and plant $\delta^{15}$N suggest that N limitation was associated with increased use of isotopically depleted N sources such as mineralized organic matter. The increases in N$_2$O fluxes with warming suggest that soil N$_2$O fluxes could serve as a positive feedback to
global warming in turfgrass. The strong influence of soil moisture on N₂O fluxes suggests that best management practices for turfgrass should optimize the tradeoff between soil moisture enhancement of urea fertilizer hydrolysis and gaseous N emissions. That is, soil moisture should be regulated so that plant use of fertilizer is maximized while gaseous N loss is minimized (e.g. Matson et al., 1998).

These results also provide in situ evidence for rapid, warming-induced C₄ weed expansion in turfgrass and suggest that other managed systems, such as agricultural crops, may experience rapid weed invasions or changes in community composition in response to warming. Since turfgrass is often associated with urban and suburban land cover, the urban heat island effect as well as climate change may exacerbate weed invasions, which would require more intensive management, e.g. herbicide application, to manage species composition.

2.7. Acknowledgements

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Chapter 3

A comparative study of lawn water budgets in southern California

3.1. Abstract

The fate of irrigation water in urban landscapes is highly uncertain, due in large part to uncertainties in urban ecohydrology. In this study, we compared irrigation rates, soil moisture, leaf-level stomatal conductance, evapotranspiration, and water budgets of three landscapes managed with different irrigation technologies and turfgrasses. The “Typical” landscape consisted of tall fescue (*Schedonorus phoenix* (Scop.) Holub), a common cool-season turfgrass species. Watering was controlled by a timer and occurred on a regular basis, representative of irrigation by most homeowners. The “Retrofitted” landscape consisted of seashore paspalum (*Paspalum vaginatum*), a warm-season species, and watering was controlled with a soil moisture sensor-based irrigation system. The “Low-impact” landscape consisted of a cool-season native sedge (*Carex*), and was equipped with a weather station-based drip irrigation system. The amount of irrigation applied was highest in the Typical landscape, and lowest in the Retrofitted landscape. There was no significant difference in soil moisture among landscapes, except in spring when the Retrofitted landscape was drier than the Low-impact landscape. The stomatal conductance of sedge was lower than the other two species. However, average annual ET
was similar among landscapes because of differences in leaf area index. Within-canopy vapor pressure deficit was extremely low (<0.6 kPa), suggesting low soil evaporative losses. The Typical landscape had the lowest irrigation efficiency (ET/applied irrigation) of 16%. Contrary to expectation, the Low-impact landscape showed only a slightly higher IE of 24%, while the highest IE of 43% was measured in the Retrofitted landscape. Runoff accounted for less than 2% of applied irrigation in each landscape. However, drainage below the rooting zone accounted for over half of applied irrigation. Our results indicated that the soil moisture sensor-based irrigation system provided the greatest water savings, and that choice of turfgrass species had little impact on the water budget. However, all three landscapes were over-irrigated, even those fitted with “smart” irrigation technology.

3.2. Introduction

Landscape water use has increasingly become of concern due to limited water resources. Turfgrass is a major landscape crop covering 2% of the U.S. surface area (Milesi et al., 2005) and may require supplemental irrigation, especially in arid and semi-arid areas such as California. According to Gleick, (2003), landscape irrigation is the dominant form of water use for residences (50-70% of household water use) as well as commercial and institutional sites (35%) in California, and is 14-30% of total municipal water use, with turf receiving most (70%) of the water. Population growth projections indicate that there will be 25 million more Californians by 2050 (State of California, 2007); therefore, landscape water use is expected to increase as a result of expanding residential and commercial development.
Because of variability in environmental conditions (weather, soil characteristics, crop needs, heterogeneity of landscapes), it is difficult for landscape managers to accurately predict plant demand, and irrigation is often in excess (Barnes, 1977, White et al., 2007). As in other regions, California has developed estimates of crop evapotranspiration (ET) to provide guidelines for irrigation. The California Irrigation Management Information System (CIMIS) is a freely available online database that uses weather data in a modified version of the Penman equation (Pruitt et al., 1977, Snyder et al., 1985) to calculate hourly reference ET, the loss of water from a standardized, well-watered crop surface which is sometimes alfalfa (ET<sub>r</sub>) but usually closely-clipped cool-season grass (ET<sub>0</sub>). The CIMIS and the Penman-Monteith ET equations result in ET<sub>0</sub> calculations that are similar (www.cimis.water.ca.gov). ET<sub>0</sub> is converted to ET for a crop with a crop-specific coefficient. Although reference ET<sub>0</sub> already represents grass ET, Meyer, (1985) determined that turfgrass can be maintained for optimum performance with crop coefficients of 0.8 and 0.6 for cool-season and warm-season grass, respectively, and 0.64 and 0.36 for minimum performance.

Automated irrigation controllers are commonly utilized by homeowners in southern California; however, these systems have been shown to increase water use by 47% (Mayer et al. 1999). Automated controllers are expected to be inefficient because they deliver irrigation irrespective of climatic or soil conditions. They can be scheduled to irrigate based on historically established ET estimates to reduce water use. Haley et al., (2007) found substantial overirrigation using homeowner scheduled automatic irrigation in Florida, and found a water savings of 30% when the controllers were set based on
historical ET. Automated controllers are now available that can adjust water application in response to changing environmental conditions, and are commonly termed “ET,” “weather-based,” or “smart” controllers. They are programmed to calculate ET₀ adjusted for landscape type based on weather and/or soil parameters, typically measured on-site. In recent years there have been numerous studies (>20) which have reported that smart controllers can promote a wide range of water and cost savings using a variety of methods, but often in comparison to historical water use (USBR, 2008 and references therein). Devitt et al., (2008) found that weather-based smart controllers reduced water consumption by 20% compared to a time-based irrigation treatment in Las Vegas; similarly, Davis et al., (2009) observed 43% water savings in Florida. Both studies found that controllers did not alter turf quality. McCready et al., (2009) found water savings of 7–30% for rain sensor-based controllers, 25–62% for weather-based controllers, and 0–74% for soil moisture-based controllers; although turf quality was impacted at the higher end of the range. Some studies have examined water savings of smart controllers relative to potential evapotranspiration (ET₀) estimates and crop coefficients for agricultural crops (e.g. Zapata et al., 2009, Kisekka et al., 2010, Migliaccio et al., 2010). However, there are few comparative studies of smart controllers in residential landscapes (Pittenger et al., 2004) and there are no reports of direct measurements of plant function or water use, or of the fate of applied irrigation water.

In addition to irrigation technology, choices of turfgrass species or cultivar may also affect water use. Tall fescue is a C₃ perennial bunchgrass that is ubiquitous in southern California. It is popular because of its lush, dark-green appearance as well as its drought
tolerance, which is conferred by the presence of an endophyte (Swarthout et al., 2009).

Evapotranspiration of cool-season grasses such as fescue is reported to be 12-47% higher than that of warm-season grasses (Feldhake et al., 1983; Huang et al., 2000), which is 2-5 mm per day as measured by weighing lysimeters (Huang et al., 2000). The C₄ grass seashore paspalum has been proposed as lower water use species for southern California. It is commonly grown as it is a dense, salt-tolerant grass (Lee et al., 2004), known for being deep-rooted and drought resistant (Huang et al., 1997). However, weighing lysimeter measurements include drainage, and it would be worthwhile to evaluate paspalum water use with direct flux measurements under well-watered conditions.

Finally, native sedges (Carex spp.) have also been proposed an alternative lawn species. It is conventionally believed that native grasses have a low transpiration rate and are recommended for planting (e.g. http://groups.ucanr.org/sonomamg/Water-wise_Gardening/). However, they are C₃ species of riparian meadows (Winward, 1986) and their transpiration rates can be quite high (up to 8.8 mm per day, Busch, 2001). In addition, their survival is linked to soil moisture and groundwater depth (Steed et al., 2003). Thus, whether native sedges have low rates of water use in landscapes needs to be assessed.

In this study, we evaluated the water use of three residential landscapes that use different approaches to water management. The landscapes consisted of 1) a “Typical” landscape with a conventional irrigation system on a timer and a cool-season, fescue lawn; 2) a “Retrofitted” landscape with a soil moisture-based irrigation system and a warm-season, paspalum lawn, and 3) a “Low-impact” with a weather station based drip irrigation
system, and a native sedge lawn. In addition to monitoring total water application, we
directly measured the components of the water budget, including stomatal conductance of
turfgrasses, total ET, runoff, and soil moisture storage. This approach provides additional
information about irrigation efficiency and the fate of irrigation water applied in excess,
which can either be stored in the soil, lost as surface runoff, or drained below the rooting
zone. Here, we define the irrigation efficiency (IE), as the ratio of ET to total irrigation
amount.

We asked (1) How did the irrigation method affect the total amount of water applied in
each landscape? (2) How did each turfgrass species perform in terms of stomatal
conductance, growth rate, and transpiration? (3) What was the water balance and
irrigation efficiency or IE (ET/applied irrigation) of each landscape? We expected that
the Typical landscape would have the highest irrigation rate, soil moisture, stomatal
conductance and the lowest irrigation efficiency. In contrast, we expected the Low-
impact to have the lowest irrigation rate, soil moisture, stomatal conductance, and highest
irrigation efficiency. For the Retrofitted landscape, we expected the lowest ET and the
highest growth rate. In general, we sought to test assumptions about how irrigation
management and species composition influence the components of the water budget and
the efficiency of irrigation, to inform urban hydrology, ecology, and landscape
management.
3.3. Methods

3.3.1. Study sites

Three landscapes were established for experimentation and demonstration at the South Coast Research and Extension Center located at 33°41’20.16”N, 117°43’24.26”W at 123 m a.s.l. in the semi-arid, Mediterranean climate of Irvine, CA. The soil type for all landscapes was loamy sand, and was 79% sand, 10% silt, and 12% clay (UC Davis Analytical Laboratory, August 2005). Grasses on the landscapes were started as sod (West Coast Turf, Palm Desert, CA). Grass areas were split into a front and back yard with a shed in between, except in the Retrofitted landscape, which contained only a front yard. The grass areas of each landscape had some non-turf groundcovers at the peripheries, including shrubs and trees. In the Typical, these species were conventionally believed to be high-water use, in the Retrofitted they were low-water use, and in the Low-impact they were native. Each landscape included a driveway and concrete sidewalk in the front yard. In the Typical landscape, the driveway was made of continuous concrete, while the other two driveways were more permeable. The Retrofitted driveway was of a mixture of concrete, flagstone, and slot drains, and the Low-impact driveway was of interlocking concrete pavers.

Tall fescue (Schedonorus phoenix (Scop.) Holub) was planted in the Typical landscape in September 2006 with an area of 131.6 m². There was a tree in the middle of the turf area. It was mowed once a week and as needed in winter, to a height of 7.6 cm. It was fertilized with 4.9 g N/m² on April 15 and September 16 in 2008 and on February 23 and June 8 in 2009 with Scotts® Turf Builder®. There was a tree in the middle of the turf
area. It was fitted with a Rain Bird 4 Station ESP Modular Series Controller and Rain Bird Matched Precipitation Rate (MPR) spray nozzles for automatic timed irrigation (Rain Bird, Azusa, CA). Until December 31, 2008, the lawn was watered 10 minutes per day, daily. It was watered 12 minutes per day on 2 days per week from January 1 – March 23, 2009, 8 minutes per day on 4 days per week from March 24 - May 7, and 12 minutes per day every other day after May 8. These rates were based on historical reference ET.

The Retrofitted landscape was planted with seashore paspalum (*Paspalum vaginatum*) in September 2006 with an area of 42.5 m$^2$. The type and schedule of fertilization was the same as fescue, except it received less (2.5 g N/m$^2$) at each fertilization event. It was mowed at the same frequency, except to a height of 2.5 cm. Irrigation was triggered by measurements of low soil moisture measured by a Watermark Electronic Module (Irrometer, Riverside, CA) connected to a Rain Bird ESP Modular Series Controller. The soil moisture threshold for triggering irrigation was ~25 kPa at 25 cm depth.

The native sedge (*Carex* spp.) in the Low-impact landscape was planted in January 2007 with an area of 54.03 m$^2$. It was not fertilized, and was mowed once in November 2008 to a height of 2.5 cm. Drip irrigation was triggered by measurements of weather conditions measured on-site using a Hunter ET System connected to a Hunter ICC irrigation controller (Hunter, San Marcos, CA).
3.3.2. Measurements

We measured ET, leaf-level stomatal conductance, soil moisture, and leaf and soil temperature on 20 sampling days throughout July 2008 – July 2009. Within each landscape, we chose five sampling locations randomly and measured each parameter at each location 3-6 times per sampling day to capture diurnal variation. ET was measured with a portable chamber method based on McLeod et al., (2004). Briefly, a relative humidity and air temperature datalogger (HOBO® Pro V2, Bourne, MA) was placed in a clear PVC chamber that was 17.8 cm in height and 28.0 cm in width for the short turfgrasses in the Typical and Retrofitted landscapes, and 24 cm in height and 31.5 cm in width for the taller sedge in the Low-impact landscape. ET was estimated by the slope of the relationship between water vapor content and time over a 30 second interval. We tested this method with both the HOBO sensor and a fast response sensor (HMP45AC, Vaisala, Helsinki, Finland), and found no significant difference in the slope of water content over time (p>0.1). Stomatal conductance was measured with a porometer (SC-1, Decagon Devices, Pullman, WA), and leaf and soil surface temperatures were measured with a mini infrared thermometer (RAYMT6, Raytek, Santa Cruz, CA). Soil moisture (0-5 cm depth) was measured with a portable soil moisture probe (TH2O/ML2x/HH2, Dynamax, Houston, TX), which we calibrated for our soil type. We measured the actual soil moisture by the difference between wet and dry weight. This was related to the meter voltage output by a third-order polynomial with an accuracy within 2.4% VWC.

The leaf-to-air vapor pressure deficit (VPD) was determined using measurements of leaf temperature and vapor pressure at the surface (0 cm height) of the grass canopy prior to
ET measurements. We also measured within-canopy grass VPD on August 26, 2009 for the Typical and Retrofitted landscapes to compare measurements above and within the canopy. Data on incoming solar radiation (S) were taken from the CIMIS station located onsite (www.cimis.water.ca.gov, Irvine station #75).

Following mowing, clippings from each landscape were collected, dried and weighed for growth rate measurements. A small area (102.6 cm², n=5) of native sedge in was clipped to the soil surface, dried, and weighed every other week for aboveground biomass measurements. We removed thatch/dead grass from these samples to estimate the amount of live vegetation.

Specific leaf area (SLA) was determined every other week by measuring leaf area on a subset of harvested fresh leaves (ImageJ software, U.S. National Institute of Health, http://rsb.info.nih.gov/ij/) and dividing leaf area by dry weight. Leaf area index (LAI) was determined by multiplying biomass production (g/cm²) by SLA. The LAI of the stubble (remaining grass following mowing) was determined by removing small areas (19.6 cm²) following mowing in May 2009 following careful removal of thatch from live vegetation.

Time-domain reflectometers or TDR sensors (CS616, Campbell Scientific, Inc., Logan, UT, USA) were placed in the plant root zone for soil volumetric water content (VWC) up to 15 cm, 20 cm, 25 cm depth for Typical, Retrofitted, and Low-impact landscapes, respectively. These depths corresponded with observed rooting depths, although for each
species roots were concentrated in the top 5 cm of soil. We also placed a soil water potential sensor (Watermark, Irrometer, Riverside, CA) below the rooting zone at 40 cm in the Typical landscape. These sensors were logged on data loggers (CR10X, Campbell Scientific, Inc., Logan, UT, USA) every 30 seconds and averaged every 30 minutes. We calibrated the TDR sensors for our soil type. The actual soil moisture was measured by the difference between wet and dry weight, which was linearly related to the delay time with accuracy within 2.7% VWC.

In addition to ET, we measured water inputs, soil water storage, and runoff, and estimated drainage by difference. Soil moisture storage was measured as the change in VWC over time. Runoff was measured by collecting all surface runoff in 0.3 x 0.3 m concrete vaults downslope of each landscape. Electronic water sensing sump pumps (Water Ace, Ashland, Ohio) transported runoff through an oscillating piston type water meter pulse flow meter (C700, Elster AMCO Water, Langley, Canada) which was logged daily (CR1000, Campbell Scientific, Inc., Logan, UT, USA). Although the area that the runoff collectors covered included some non-turf areas of the landscapes, we assumed that all runoff came from turfgrass, which was the largest component of each landscape. Therefore, the recorded values may be slight over-estimates of turf runoff. Precipitation data was obtained from a CIMIS station located onsite (www.cimis.water.ca.gov, Irvine station #75). Irrigation was measured on a weekly basis between March 14, 2008 and July 22, 2008 by distributing 18 cups evenly every ~1 m² on the Typical lawn. On the other two lawns, 3 cups were evenly spaced. All cups were filled with 2 cm of mineral oil to prevent evaporation. In three sprinklers of the Typical landscape, we used the electrical
conductivity method to estimate the amount of water lost by evaporation in the water spray by measuring increases in salinity (McLean et al., 2000).

3.3.3. Data analyses
Measured ET (mg/m²/s) was filtered to remove points where S was less than 10 Wm⁻² and VPD was less than 0.1 kPa. Filtered ET was modeled as a function of VPD and S linearly, similar to Granier et al., (1996).

\[
ET = y_0 + a \cdot \ln(\text{VPD}) + b \cdot S
\]  

(1).

Residuals were plotted against measured soil moisture (0-5 cm) at the site of ET measurement to evaluate the influence of soil moisture on ET. We also calculated total conductance to vapor (gᵥ) by

\[
g_v = \frac{ET \cdot p_a}{\text{VPD}}
\]  

(2).

where \( p_a \) is air pressure, and compared these values to stomatal conductance measurements.

Drainage was estimated as a residual of the water budget, the other components of which were directly measured:

\[
D = P + I - ET - R - \Delta SM
\]  

(3)

where D, P, I, R, \( \Delta SM \) stand for subsurface drainage, precipitation, irrigation, runoff, and change in soil moisture storage, respectively. A two-hour running average of the soil water potential was performed for smoothing.
All statistical analyses were performed on SAS 9.1.3 software (SAS Institute Inc., Cary, NC). The influence of landscape and time on plant and soil properties was evaluated with repeated measures analyses of variance (ANOVA) using the General Linear Model. For stomatal conductance, we removed points where S was less than 10 Wm$^{-2}$ and VPD was less than 0.1 kPa, and we compared seasonal rather than daily values, since it was not possible to take measurements simultaneously on all landscapes. Post-hoc tests were conducted using the Tukey Standardized Range Test. Two-sided paired t-tests on Excel (Microsoft Corp., Redmond, WA) were used to compare measured stomatal conductance and g$_v$. For all analyses, p < 0.05 was considered significant, and p < 0.1 was considered marginally significant.

3.4. Results

Climatic conditions during the study period are shown in Figure 1. Maximum daytime temperature was 32.0°C in April, while the minimum was 8.3°C in December. Average daily leaf-to-air VPD above the grass canopies varied between 0.2 - 3 kPa and showed a seasonal trend. Episodes of high daytime temperature corresponded with Santa Ana winds, a reversal in the direction of the sea breeze due to high pressure in inland deserts. Average daytime S declined from highs of ~475 Wm$^{-2}$ in spring and summer to ~300 Wm$^{-2}$ in winter. The precipitation was 214 mm during the study period, and occurred only in winter and spring months. There was an approximately 8 month period without precipitation, when irrigation was the only source of water to the plots. Within-canopy VPD was <0.6 kPa on August 25, 2008.
Figure 1 – Average values of environmental variables during the study period: (a) daytime temperature, (b) daytime above-canopy VPD, (c) daytime solar radiation, and (d) precipitation.
Leaf temperatures followed a seasonal cycle corresponding with air temperature. The temperature of the fescue leaves in the Typical landscape were greater than air temperature by $2.1 \pm 0.33^\circ C$, except between November and early February when they were $2.5 \pm 0.5^\circ C$ lower (Figure 2a). Similarly, the temperature of the paspalum leaves in the Retrofitted landscape were $2.6 \pm 0.35^\circ C$ greater than air temperature, except between November and December when they were $5.7 \pm 0.4^\circ C$ lower (Figure 2b). The temperature of the sedge leaves in the Low-impact landscape were $1.4 \pm 0.35^\circ C$ greater than air temperature, except in November when they were $3.2 \pm 0.4^\circ C$ lower (Figure 2c).

**Figure 2** – Time series of leaf and air temperatures measured simultaneously at the sites for each species with SD for (a) fescue (b) paspalum and (c) native sedge. Asterisks (*) indicate when leaf temperature exceeded air temperature, and bold asterisks (**) indicate when air temperature exceeded leaf temperature.
The VWC in the rooting zone was initially similar in all three landscapes (~18%), and increased slightly during summer in the Typical and Low-impact landscapes by about 1% and 3%, respectively (Figure 3), but declined in the Retrofitted landscape by about 4% (p<0.0001). There was no difference in soil moisture among landscapes until May 22, 2009, when the Retrofitted landscape became marginally drier than the Low-impact landscape (Figure 3; p=0.0607).

Figure 3 – Time series of soil volumetric water content (VWC) with SE. There were no differences in the soil moisture of landscapes until May 2009, when the Retrofitted landscape became marginally drier than the Typical landscape (p=0.0607).
The observed upper limit in VWC for the surface soils (0-5 cm) was \(~44\%\) (not shown), which corresponds to the porosity for our loamy sand soil type (Rawls et al., 1982). These surface soils were always saturated or near saturation (within \(10\pm6\%\)). There were sharp increases in soil moisture during irrigation events, as shown for VWC recorded in the root zone of the Typical landscape (Figure 4). The water potential sensor placed below the root zone at 40 cm also showed increases corresponding with irrigation events.

**Figure 4** – Time series of soil volumetric water content (VWC) in the root zone, and water potential below the root zone.
Stomatal conductance did not change seasonally for any species (Figure 5, Table 1; \( p > 0.1 \)), and there was no difference in stomatal conductance of fescue versus paspalum (\( p > 0.1 \)). However, native sedge had lower stomatal conductance than fescue (\( p = 0.0153 \)) and paspalum (\( p = 0.0027 \)).

![Figure 5](image.png)

**Figure 5** - Time series of seasonal averages of diurnal stomatal conductances for each species with SD.

<table>
<thead>
<tr>
<th>Species</th>
<th>Stomatal conductance (mmol/m²/s)</th>
<th>( g_v ) (mmol/m²/s)</th>
<th>SLA (cm²/g)</th>
<th>One-sided LAI</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tall fescue</td>
<td>139 ± 68</td>
<td>132 ± 102</td>
<td>232 ± 33</td>
<td>6.8 ± 2.4</td>
</tr>
<tr>
<td>Seashore paspalum</td>
<td>160 ± 101</td>
<td>102 ± 78</td>
<td>313 ± 46</td>
<td>5.9 ± 1.8</td>
</tr>
<tr>
<td>Native sedge</td>
<td>115 ± 46</td>
<td>126 ± 84</td>
<td>150 ± 32</td>
<td>8.7 ± 3.9</td>
</tr>
</tbody>
</table>
Figure 6 - Time series of (a) growth rate for tall fescue and seashore paspalum (b) aboveground biomass for native sedge.

The growth rate of paspalum and fescue were similar, except in spring and summer when paspalum growth rate increased (Figure 6a). Aboveground biomass for native sedge between August and November 2008 did not change significantly (Figure 6b; p>0.1) and was 340.7 ± 242.9 g/m² (SD). Following the mowing of native sedge in November 2008, biomass declined to 102.6 ± 63.9 g/m² and did not change significantly over time (p<0.0001). There was no trend in SLA over time for any of the species (not shown; p>0.1) and SLA differed among all species (p<0.05). SLA was highest in paspalum at
313 ± 46 cm²/g and intermediate for fescue at 232 ± 33 cm²/g. SLA was lowest in the native sedge at 150 ± 32 cm²/g for flat leaves and 58 ± 36 cm²/g for triangular leaves, which were present following spring. The one-sided LAI of stubble measured in May was 6.8 ± 2.4 for fescue and 5.9 ± 1.8 for paspalum. The LAI of the mowed biomass added a trivial amount of leaf area to the stubble LAI (0.2 ±0.1 for fescue and 0.6 ± 0.3 for paspalum). The one-sided LAI for the native sedge was 8.7 ± 3.9.

Figure 7 – Time series ET measurements of each species measured using the chamber technique, and ET₀ measurements calculated by CIMIS.

ET did not vary among landscapes, and showed similar seasonal trends in all three landscapes (Figure 7). ET was highest in July 2009 at ~1.5 mm/day, and was lowest between November and February at ~0.3 mm/day. Integrated ET from July 2008-July 2009 was 194.3 ± 33.3 mm, 210.9 ± 35.4 mm, 247.9 ± 46.9 mm for the Typical,
Retrofitted, and Low-impact landscapes, respectively. Chamber ET measurements were much lower than calculated CIMIS ET0 (18 ± 6% of CIMIS ET0).

VPD and S were significantly correlated with ET and explained 50-59% of variation when modeled with equation (1). The model coefficients are shown in Table 2. Model residuals were not related to VWC at 0-5 cm soil depth (p>0.1). Measured stomatal conductance was similar to calculated $g_v$ for fescue (139 ± 68 mmol/m²/s vs. 132 ± 102 mmol/m²/s, p>0.1), while stomatal conductance was slightly higher than $g_v$ for paspalum (160 ± 101 mmol/m²/s vs. 102 ± 78 mmol/m²/s, p<0.0001) and slightly lower than $g_v$ for native sedge (115 ± 46 mmol/m²/s vs. 126 ± 84 mmol/m²/s; p<0.05).

Table 2: Mean model parameters (±1 standard error) describing canopy ET (mg/m²/s) in relation to light (W/m²) and vapor pressure deficit (kPa). Model residuals were not related to soil moisture at 0-5 cm depth.

<table>
<thead>
<tr>
<th>Species</th>
<th>$R^2$</th>
<th>$y_0$</th>
<th>$a$</th>
<th>$b$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tall fescue</td>
<td>0.52</td>
<td>4.5 ± 2.1</td>
<td>5.8 ± 1.0</td>
<td>0.0294 ± 0.0031</td>
</tr>
<tr>
<td>Seashore paspalum</td>
<td>0.59</td>
<td>-0.5 ± 2.5</td>
<td>6.2 ± 1.1</td>
<td>0.0412 ± 0.0040</td>
</tr>
<tr>
<td>Native sedge</td>
<td>0.50</td>
<td>7.5 ± 2.5</td>
<td>6.6 ± 1.2</td>
<td>0.0300 ± 0.0039</td>
</tr>
</tbody>
</table>

Irrigation was 770 ± 100 mm, 313 ± 21 mm, and 550 ± 70 mm for the Typical, Retrofitted, and Low-impact landscapes between March 14, 2008 and July 22, 2008 (Figure 8). Spray loss via evaporation from sprinkler head to the landing point ranged from 0.6-1.4% in the Typical sprinklers, and was assumed to be negligible in the calculated water budget. The Retrofitted landscape had the highest IE (43 ± 9%), the Low-impact was intermediate (24 ± 8%), and the Typical landscape was lowest (16 ± 4%). The proportion of irrigation lost to drainage was similar in both the Typical and Low-impact landscapes (82 ± 17% vs. 76 ± 18%), and was 57 ± 12% in the Retrofitted
landscape. Runoff was $1.6 \pm 0.2\%$, $0.2 \pm 0.0\%$, and $0.3 \pm 0.0\%$ for the Typical, Retrofitted, and Low-impact landscapes. Soil water storage did not change significantly in the Typical and Low-impact landscapes during this period, but in the Retrofitted landscapes, soil water loss was 7.1 mm or 2.3% of irrigation.

**Figure 8** - The water budget as a percentage of irrigation applied to each landscape from March 14, 2009 to July 22, 2009. Amount of irrigation applied and precipitation for this time period is indicated beneath each pie chart.

Precipitation was 0.6%, 1.6%, and 0.9% of irrigation for the Typical, Retrofitted, and Low-impact landscapes, respectively. In the Retrofitted landscape, measurable runoff occurred only on one day, March 22, with a rainfall event of 3 mm. For the other rain events, daily rainfall was less than 0.8 mm and there was no measurable runoff. The majority (65.8%) of runoff from the Low-impact landscape was from the March 22 rainfall event, and 11.4% of runoff from the Typical landscape was due to this event. In contrast, there was measurable runoff from the Typical landscape after every rain event.
Most of the runoff from the Low-impact landscape (76.7%) was from rain events rather than irrigation events, while 20.3% of the runoff from the Low-impact landscape was from rain events.

3.5. Discussion

We found low irrigation efficiencies (16-44%), with a high proportion of water loss as drainage (16-43%) in all three landscapes (Figure 8). Differences in IE were due to amount of irrigation applied, since landscapes had similar rates of ET. Unexpectedly, the C₄ paspalum and native sedge transpired at the same rate as the C₃ fescue in the Typical landscape. Stomatal conductance was similar in fescue and paspalum, but lower in native sedge, which compensated with a higher LAI and resulted in similar ET. Soil moisture was near saturation in all landscapes and did not influence ET. In general, when drainage losses are considered, there was over-irrigation in all three landscapes, even those fitted with “smart” irrigation systems.

3.5.1. The role of irrigation method

The Retrofitted landscape received the least amount of water, 313 ± 21 mm for the March 14 – July 22, 2009 period, contrary to our hypothesis. The Typical landscape received more than twice that amount, or 770 ± 100 mm. The soil moisture-based irrigation controller in the Retrofitted landscape was far more effective at reducing water application than the automatic timed controller, as has been in other studies (McCready et al., 2009, Devitt et al., 2008). However, it was surprising that the drip irrigation by the weather station-based controller in the Low-impact landscape was only slightly lower
than the Typical landscape at $550 \pm 70$ mm. This implies that soil moisture-based irrigation may be more sensitive to plant water needs than the weather station based systems, which utilize atmospheric conditions rather than plant available water. Our results concur with Pittenger et al., (2004), who found that smart irrigation systems do not necessarily reduce water use.

As a result of the similarity in watering rates, soil moisture in the root zone was similar in the Typical and Low-impact landscapes (Figure 3). However, the Retrofitted landscape experienced a period of reduction in soil moisture due to an imbalance between summer evaporative demand and ET (Figure 3).

3.5.2. The role of turfgrass species

It is a commonly assumed that warm-season turfgrasses will transpire less than cool-season turfgrasses (Meyer et al., 1986), but this was not the case in our study. We found higher a growth rate in the warm-season grass than the cool-season grass in the warmer spring and summer months as hypothesized, similar to Robins, (2010). However, ET did not differ among landscapes (Figure 7). Our modeling results showed that ET was not related to surface soil moisture, where most roots were concentrated (Table 2; $p>0.1$), as surface soil was always saturated or near saturation. Thus, it is likely that our measured rates of stomatal conductance represent maximum values, which may not differ between fescue and paspalum (Figure 5). It is conventionally believed that native grasses have a low transpiration rate, and in fact, leaf-level stomatal conductance was lowest for the
native sedge (Figure 5), but this did not scale up to lower plot-level ET (Figure 7), as sedge had a higher LAI (Table 1).

Stomatal conductance and $g_v$ were similar in fescue, and only slightly different in paspalum and native sedge (within about 58 mmol/m$^2$/s). With negligible boundary layer resistance and soil evaporation, this corresponds to an active LAI of about 1, although our one-sided LAI was $\geq 5.9$. However, the leaf-to-air temperature differences (Figure 2) and low within-canopy VPD (<0.6 kPa) indicate a high boundary layer resistance, as would be expected for a lawn (Schulze et al., 2005). Thus, the active leaf area index is probably higher than 1, though perhaps not as high as estimated by (Allen, 1998), who assumed that the active LAI is half the total LAI.

Our values of stomatal conductance for fescue were within previously observed laboratory values, although lower than well-watered fescue under laboratory conditions (~400 mmol/m$^2$/s) (Zhao et al., 2008, Swarthout et al., 2009). Stomatal conductance of native sedge was within the reported range for Carex species grown in drained soils (Busch et al., 1998, Busch, 2001). We did not find previous estimates of stomatal conductance in paspalum. Although we did not find evidence of lower water use in paspalum, contrary to previous studies (Feldhake et al., 1983; Huang et al., 2000), the C$_4$ photosynthetic pathway supported a higher growth rate than the C$_3$ fescue with a similar level of stomatal conductance (Figures 4,5), as expected for a warm season species.
Our measured SLA was within the reported range (Volk et al., 2000; Bijoor et al., 2008, Liang et al., 2009) for sedge and fescue; previous values were not available for paspalum. Our values of LAI are similar to Lee (Lee, 2008), who reported LAI up to 9 for various turfgrasses. Our estimated LAI of mowed material was similar to Bijoor et al., 2008), although these values depend on mowing height.

3.5.3. ET and water balance

Soil evaporation probably played a small role in ET and therefore in the total water budgets. The high VPD inside the short grass canopies suggests that soil evaporation was low, as within-canopy VPD did not exceed 0.6 kPa even during a hot summer afternoon in August. Our measured ET values are similar to those reported for irrigated urban forests in this region (Pataki et al., in press). ET measurements were spatially and temporally variable, although with less day-to-day variability than CIMIS ET0 (Figure 7). Chamber ET was generally lower than previous estimates obtained from minilysimeters (Beard et al., 1989). The CIMIS system uses the Pruitt and Doorenbos (1977) reference ET0 equation which was calibrated using a weighing lysimeter in unstressed, cool season grass. A number of studies have found that calculated reference ET agrees well with weighing lysimeter data (e.g. Allen et al., 1989, Ventura et al., 1999, Howell et al., 2000, Wright, 2000), although other studies have shown that turfgrasses can transpire less (Meyer, 1985), and irrigation at 20% of reference ET has been shown to result in enhanced deep rooting in fescue (Fu et al., 2007). CIMIS ET0 measurements were much higher (82 ± 6%) than our chamber ET measurements on most days (Figure 7). There was an exception during a day with low S, when CIMIS measurements were 229 ± 89% lower
than our ET measurements (0.15 mm/d vs. 0.4-0.6 mm/d). The CIMIS method is known to be sensitive to measured solar radiation (Itenfisu et al., 2003). It is not surprising that in general, lysimeter measurements are higher than our chamber-based ET measurements. Lysimeters have the potential to overestimate ET due to heating and cracks in soil along the lysimeter wall (Rana et al., 2000), but more importantly, because they include drainage losses, which we conclude were a very important component of the water budget in our study.

Our inferred rates of drainage are supported by the below-root water potential data, which shows rapid increases in water potential coincident with irrigation events (Figure 4). Other studies have also shown high rates drainage in loamy sand soils (Prunty et al., 1998; Roy et al., 2000, Ochoa et al., 2007). Loamy sand has a high saturated hydraulic conductivity (Maidment, 1993), which increases in the presence of turfgrass (Roy et al., 2000). Saffigna et al., (1977) used drainage lysimeters for potato crops in loamy sand soils and showed that more than half of the total water inputs were lost to drainage using conventional irrigation scheduling.

The Retrofitted landscape had the highest irrigation efficiency (43 ± 9%), followed by the Low-impact (24 ± 8%), and the Typical landscape (16 ± 4%) (Figure 8). These differences were largely due to the amount of irrigation applied, since ET rates were similar across landscapes. Runoff was not an important component of the water budget in any of the landscapes (Figure 8), but it was highest in the Typical landscape, possibly due to the concrete hardscape. Runoff may have been higher and drainage lower if the
experiment had been conducted in finer textured soils, which should be evaluated with additional experiments.

3.6. Conclusions

We evaluated the water budgets of three landscapes that utilized varying degrees of water conservation measures. The Retrofitted landscape, which was equipped with a soil moisture sensor-based irrigation system and was planted with a warm-season grass, had the highest IE (44%) The Typical landscape was equipped with a standard timer-based irrigation system and a cool season fescue lawn, and had the lowest IE (16%). Surprisingly, the Low-impact landscape had only a slightly higher IE of 24%, despite a weather station-based drip irrigation system and a native sedge lawn. A priori, the Low-impact landscape was expected to have the highest IE, as it utilized the most recommended water conservation measures. Differences in IE were largely due to amount of irrigation applied, since landscapes had similar rates of ET. In this study, irrigation technology was much more important in influencing landscape water budgets than choice of turfgrass species. However, even “smart” irrigation systems resulted in large drainage losses. In addition, our estimates of ET were much lower than CIMIS ET₀, suggesting that CIMIS predictions may not apply to turf in well-drained soils. This study should be replicated in other soil, climatic, and landscape conditions, but shows the important of direct measurements of the components of water balance for determining effective irrigation rates and water conservation strategies.
3.7. Acknowledgements

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Chapter 4

The application of $\delta^{18}O$ and $\delta^D$ for understanding water pools and fluxes in a *Typha* Marsh

4.1. Abstract

The $\delta^{18}O$ and $\delta^D$ composition of water pools (leaf, root, standing, and soil water) and fluxes (transpiration, evaporation) were used to understand ecohydrological processes in a managed *Typha latifolia* L. freshwater marsh. The isotopic composition of standing water in the marsh became isotopically enriched over time, consistent with progressive evaporation. We utilized these measurements to assess pathways of water loss, and determined that transpiration accounted for most (82 ± 2%) of the cumulative evapotranspiration (ET) from March to June. Evaporation accounted for 3.6% of the total water loss, transpiration accounted for 15%, and subsurface drainage accounted for the majority, 81.4%. The isotopic composition of modeled leaf water was similar to observed leaf water, indicating that plants were transpiring in isotopic steady state. We also measured the isotopic composition of water vapor within the canopy near the standing water or soil surface as well as above the canopy, and found a vertical gradient in isotopic composition and water vapor content sufficient for distinguishing between evaporation and transpiration (T) on 38% of sampling days. For these periods, we found that mean T/ET was between 38 ± 67% and 96 ± 67%, but the estimated error was
relatively high (>37%). The greatest uncertainty was associated with periods when atmospheric water vapor was dominated by non-local, background sources (>90%). However, independent estimates of T/ET using eddy covariance measurements yielded similar mean values during the *Typha* growing period. Hence, the isotopic mass balance of water vapor yielded reasonable results, but the mass balance of standing water provided more definitive estimates of water losses.

### 4.2. Introduction

Water balance plays an integral role in the functioning of wetlands, influencing ecology, water supply and quality, and carbon storage (Lafleur, 2008, Rocha *et al.*, 2010), and its understanding is important for effective wetland management. There is a long-standing debate concerning many aspects of water balance components, such as the relative importance of transpiration to evaporation. Many studies have shown that most wetlands have lower evapotranspiration (ET) than would be expected from open water under similar conditions (Linacre *et al.*, 1970, Idso *et al.*, 1988, Lafleur, 1990, Burba *et al.*, 1999). However, other reports show that the presence of wetland reed vegetation increases ET (Price, 1994, Herbst *et al.*, 1999, Pauliukonis *et al.*, 2001, Acreman *et al.*, 2003, Goulden *et al.*, 2007). In addition, it is believed that ET usually represents the dominant water loss from ecosystems (Drexler *et al.*, 2004), but few studies have examined the complete water budget of wetlands, including groundwater flows (e.g. Owen, 1995, Hunt *et al.*, 1996).
The stable isotopic composition ($\delta^{18}O$ and $\delta D$) of ecosystem water pools such as leaf, root, and surface water and atmospheric water vapor may help resolve some of the uncertainties associated with wetland ecohydrologic processes. Variations in the isotopic composition of water can provide information about water sources and transport within ecosystems. Fractionating processes lead to distinct isotopic signatures of water pools (Lai et al., 2006, Welp et al., 2008). While this information has many applications, there are limited datasets on temporal variability in water pools other than precipitation (e.g., IAEA/WMO, 1981, Rozanski et al., 1993, Johnson et al., 2004), in part because it is usually necessary to extract water from plant or soil material (West et al., 2006) or condense atmospheric vapor (Helliker et al., 2002) for conventional measurements.

Differences in the isotopic composition of pools and fluxes can be used to construct the isotopic mass balance of the standing water in a wetland to understand hydrologic dynamics (Hunt et al., 1996, Hunt et al., 1998, Clay et al., 2004, Sikdar et al., 2009, Nyarko et al., 2010). Isotopic mass balance has the ability to quantify both transpiration (T) and evaporation (E), which both transport large amounts of water to the atmosphere, but are experimentally difficult to distinguish. Water subject to evaporation becomes enriched, while drainage and transpiration fluxes are believed to be non-fractionating (Gonfiantini, 1986, Dawson et al., 1991, Dawson et al., 2002). Thus, in a system where evaporation is the only fractionating outflow, it is possible to estimate evaporative losses when combined with direct measurements of ET.
In addition, several studies have highlighted the potential of stable isotopes to detect transpiration in canopy water vapor in a variety of ecosystems (Brunel et al., 1992; Yakir et al., 1996; Brunel et al., 1997; Wang et al., 2000). By understanding the $\delta^{18}O$ and $\delta D$ isotopic composition of water vapor that is transpired or evaporated, as well as the remaining or “background” vapor, it can be possible to obtain quantitative estimates of the contribution of each component, provided that they are isotopically distinct (Zhang et al., 2010). The isotopic composition of water evaporated from soil or water bodies may be isotopically lighter than transpired water vapor, which is similar to plant source water, assuming steady state. Studies of the isotopic composition of canopy water vapor have previously shown that transpiration can be the dominant component of ET in forests (Moreira et al., 1997, Harwood et al., 1999), savannah woodlands (Yepez et al., 2003), and orchards (Williams et al., 2004). However, evaporative fluxes may play a large or even dominant role in wetlands with standing water.

The isotopic composition of leaf water is important for understanding the isotopic composition of transpiration. It can be predicted using a simple model based on evaporative enrichment (Craig et al., 1965). This model has been modified to describe progressive enrichment along the length of the elongated leaves such as grasses using the string-of-lakes model (Gat et al., 1991a, Helliker et al., 2000). However, two effects may cause discrepancies between measured and modeled leaf water. The “Péclet effect” is a result of advection and back diffusion between less enriched water in veins and enriched water at evaporative sites (Barbour et al., 2004). Gan et al., (2002) emphasized the need to include Péclet effects when applying the string-of-lakes model to leaves. Secondly,
leaves may not be transpiring in isotopic steady state if leaf water pools change in size (Flanagan et al., 1991; Farquhar et al., 2005, Lai et al., 2006, Seibt et al., 2006, Lee et al., 2007). It is possible to take both of these effects into account when modeling leaf water, which is also useful for a variety of studies including physiological and paleoclimatic studies of the isotopic composition of plant biomass (Epstein et al., 1977, Edwards et al., 1985, Edwards et al., 1988, Roden et al., 1999), and measurements of the oxygen isotope composition of CO₂, which equilibrates with leaf water (Farquhar et al., 1993).

In this study, we measured the isotopic composition of leaf water, root water, soil water, standing water, and water vapor over the course of a growing season in a managed Typha latifolia L. freshwater marsh in Orange County, California, where eddy covariance measurements of ET were available. A previous study of direct flux measurements at this site has shown that despite a large area of standing water, evaporation rates were low, and ET was dominated by transpiration (Goulden et al., 2007). The previous study also suggested that plants were deeply rooted, even though high soil moisture availability was expected to promote surficial roots. We followed up on these findings with the questions: 1) Are ecosystem water pools and fluxes isotopically distinct?, 2) Is there evidence of shallow rooting in Typha using isotopic methods? 3) Is there support for a Péclet effect and/or isotopic non-steady state transpiration in Typha?, 4) What is the water balance of study system? 5) What is the proportion of transpiration in ET (T/ET), and how do estimates based on the isotopic mass balance of standing water (T/ET_{SW}) and
the isotopic mass balance of water vapor (T/ET<sub>WV</sub>) compare with an eddy covariance-based approach (T/ET<sub>EC</sub>)?

We hypothesized that plant transpiration would be primarily in isotopic steady-state, which has been shown for short canopies (Yakir & Wang 1996, Welp et al., 2008). We expected that transpiration would dominate ET and that plants would be deeply rooted. Consequently, we expected that a small percentage of total marsh water loss would be due to evaporation, and that the largest water losses would be transpiration and subsurface drainage. In general, we sought to assess the extent to which isotope mass balance studies can inform wetland ecology and ecohydrology.

4.3. Materials and Methods

4.3.1. Study site

This research was conducted at the San Joaquin Freshwater Marsh Reserve (SJFM) in Orange County, California, which is 3 m above sea level and 8 km northeast of the Pacific Ocean (33° 39’ 44.4” N, 117° 51’ 6.1” W). SJFM is located on the campus of University of California, Irvine, where it is protected and managed as part of the University of California’s Natural Reserve System. The climate at SJFM is Mediterranean, and rainfall occurs primarily in the winter and spring months. The average annual temperature (1960–2008) is 16.4 ± 0.8°C and average annual precipitation is 27.0 ± 14.5 cm (http://www.wrcc.dri.edu, Newport Beach Harbor station #046175). SJFM is dominated by a dense stand of cattail (Typha latifolia L.), with a few patches of bullrush (Scirpus californicus C.A. Mey. Steud.) and willows (Salix spp.) at
the periphery, and it contains a large amount of standing litter. The hydrology of the SJFM is managed. In winter, SJFM is flooded to a depth of ~1 m with water pumped from the nearby San Diego Creek, which is composed almost entirely of urban runoff. Following the initial flooding event(s), no additional water is added except a relatively small amount via precipitation (10.5 cm during our 2007 study year); consequently, SJFM loses standing water in the following months by ET and subsurface drainage. There is no standing water left by midsummer. We recorded marsh water height with a staff gauge at one location at weekly or shorter intervals, which was assumed to be representative as basemaps showed variations in depth <0.3 m.

The site was instrumented with an eddy covariance tower, and water, CO₂ and energy exchange measurements have been previously reported (Goulden et al., 2007, Rocha et al., 2008a, Rocha et al., 2008b, Rocha et al., 2009, Rocha et al., 2010). The eddy covariance tower provided direct measurements of ET, and a similar tower fitted at a nearby dry pond helped develop estimates of T/ET_EC over the 2007 growing season. Instrumentation and flux calculations for these towers have been discussed in detail previously. Briefly, the Net Ecosystem Exchange (NEE) and ET were calculated from the average half hour covariance between vertical wind fluctuations measured by a 3D sonic anemometer (CSAT3, Campbell Scientific, Logan, UT) and the mole fractions of CO₂ for NEE and H₂O for ET measured by a closed path Infrared Gas Analyzer (IRGA; Model 7000 or 6262, Li-Cor Inc., Lincoln, NE). Fluxes were rotated to the mean wind streamlines, and corrected for underestimation of high frequency flux from to tube attenuation and the density effect due to sensible and latent heat transfer. Turbulent
fluxes of sensible and latent heat captured 78–80% of the available energy at each of the sites, which is consistent with energy budget closure observed for other eddy covariance studies (Wilson et al., 2002).

4.3.2. Sample collection

We conducted intensive diurnal field campaigns to sample water pools approximately fortnightly from May through October 2007, which corresponded with the Typha growing season. In 2008, sampling days were weekly or fortnightly from January to March, a period when standing water was present, but live vegetation was sparse or absent.

On sampling days, water vapor was collected near the eddy covariance tower at two levels above the standing water or soil: within-canopy (0.1 m) and above-canopy (4 m), by condensing water vapor through a cryogenic trap at a flow of 450 cc/min for 90 minutes based on Helliker et al., (2002). Using this method, we were able to obtain ~0.5-1 mL of water vapor during sampling events. Over the course of each sampling day, we collected water vapor 3 times at each height (6 total vapor collections). Air temperature and humidity were recorded at 5 second intervals within ~3 cm of water vapor collection (HOBO® Pro V2 Temperature/Relative Humidity datalogger, Bourne, MA). From September 2007 to March 2008, vapor was collected simultaneously at a nearby beach for comparison to marsh vapor.
On each sampling day, leaves, roots, standing water or soil were collected for measurement of the $\delta^{18}$O and $\delta$D composition of water. All samples were collected within about 10 meters of the vapor collection. To account for progressive isotopic enrichment along the leaf axis, 5 cm segments were collected from the base, middle, and top of leaves three times over the course of the day ($n=5$ for each section of leaf). Leaf samples were collected once in the middle of each vapor sampling event (about 45 minutes into the vapor sampling). We collected root samples from five plants at midday on each sampling day during the 2007 growing season. Standing water from the top centimeter was collected on a weekly basis and on sampling days at five locations. Surface soil samples (0-5 cm depth) were collected at five locations during the period without standing water. Leaf, water, soil, and root samples were immediately placed in screw cap vials after collection, sealed with parafilm, and stored frozen. Prior to isotopic analysis, water was extracted from leaves, soils, and roots using cryogenic vacuum distillation (West et al., 2006).

Leaf water content (LWC), stomatal conductance, and specific leaf area (SLA) were also measured. Leaves were collected for LWC measurements ($n=50$) on each sampling day. Leaves were measured on-site for wet weight after clipping. Leaf biomass was oven-dried at 70°C for at least 48 hours and measured for dry weight in the lab. LWC was determined as the difference between fresh and dry leaf weight. Stomatal conductance was measured with a porometer (SC-1, Decagon Devices, Pullman, WA) on each sampling day. Measurements were taken during midday on five leaves in five equally spaced sections of the leaf from base to top. SLA was measured twice during the study.

4.3.3. Isotope analysis

All water samples were analyzed at the University of California, Irvine for oxygen and hydrogen isotope ratios ($\delta^{18}$O and $\delta^D$) using a thermal conversion elemental analyzer (TC EA, Thermofinnigan, San Jose, CA) coupled to an Isotope Ratio Mass Spectrometer (Delta Plus XP, Thermofinnigan, San Jose, CA) as per Gehre et al., (2004). Oxygen and hydrogen isotope measurements were expressed in common $\delta$ notation. $\delta^{18}$O and $\delta^D$ was referenced to V-SMOW with a precision of 0.15‰ and 1.0‰ (SD), respectively.

4.3.4. Statistical analyses

All statistical analyses were performed on SAS 9.1.3 software (SAS Institute Inc., Cary, NC). Repeated measures analyses of variance (ANOVA) were performed for time-series of plant, surface water and water vapor properties using the General Linear Model for Analysis of Variance. Post-hoc tests were conducted using the Tukey Standardized Range Test. Least squares linear regressions were used to assess changes in stomatal conductance, leaf isotopic composition and water content over time. For all analyses, $p < 0.05$ was considered significant, and $p < 0.1$ was considered marginally significant.
4.3.5. Calculation of marsh water balance and T/ETsw

We used a model of isotopic mass balance for estimating evaporation from standing water by Gonfiantini, (1986). We began our analysis following the transfer of some marsh standing water to another pond on March 23, 2007. This caused a drop in standing water height by almost 0.15 m. After this date, marsh water was undisturbed and water losses were assumed to be dominated by evaporation (E), transpiration (T), and subsurface drainage (D). We modeled the proportion of evaporative water loss since the last sampling day according to Gibson, (2002), or E/(E+T+D). According to Gibson (2002), for a lake in which inflow (I) equals zero and evaporation (E) and non-fractionating outflows (Q) such as transpiration and subsurface drainage are not negligible, the mass balance of the pond is represented as

\[ \frac{dV}{dt} = -Q - E \] (1), and

the isotope balance is represented as

\[ \delta_L = \delta^* - (\delta^* - \delta_0)m^{E/(E+Q)} \] (2),

where \( \delta_L \) is the isotopic composition of the lake, \( \delta^* \) is the limiting isotopic composition under local meteorological conditions as defined by Gat and Levy (1978) and Gat (1981), and \( m \) is a measure of the rate of approach to isotopic steady state as defined by Welhan and Fritz (1977) and Allison and Leaney (1982). For each time step, we solved for \( E/(E+Q) \) which equals \( E/(E+T+D) \). The depth of water loss on a particular day \( (E+T+D) \) was calculated as the difference between initial depth on March 23 (0.7 m), and depth on that day \( (H_t) \):

\[ 0.7 - H_t = E+T+D \] (3).
This was multiplied by \( E/(E+T+D) \) from Equation (2) to obtain depth of water lost via evaporation (E):

\[
E+T+D \cdot \frac{E}{E+T+D} = E \quad (4).
\]

For the marsh water balance, E was subtracted from depth of ET loss obtained via the eddy covariance method to obtain T. Depth of ET loss was subtracted from depth of water loss (E + T + D) to obtain D.

E was divided by depth of ET loss to obtain E/ET. Finally, T/ET for the time step was calculated as follows.

\[
T/ET = (1 – E/ET). \quad (5).
\]

4.3.6. Calculation of T/ET_{WV}

We used the model by Craig et al., (1965) to estimate the isotopic composition of evaporated vapor (\( \delta^{18}O_e \) and \( \delta^D_e \)) from the marsh surface, which was described by Moreira et al., (1997):

\[
R_E = \left( \frac{1}{\alpha_K} \right) \frac{R_S / \alpha^* - R_A h}{(1 - h)} \quad (6).
\]

where \( R_E \) is the molar ratio of heavy to light isotopes of \( ^{18}O \) or D of the evaporated water from the surface, \( R_S \) the molar ratio of the liquid water at the surface, and \( R_A \) is the molar ratio of the atmospheric vapor above the surface. \( \alpha^* \) is the temperature dependent equilibrium fractionation factor (Majoube, 1971), and \( \alpha_K \) is the kinetic fractionation factor for molecular diffusion in air. We used the kinetic fractionation factors of 1.021 (21‰) for oxygen and 1.011 (11‰) for hydrogen, which include the effects of a boundary layer (Cappa et al., 2003). h is the relative humidity of the air.
To understand the isotopic composition of transpiration, we needed to test whether transpiration was in isotopic steady state. To do this, we estimated leaf water isotopic composition using a model of steady state behavior. If modeled *Typha* leaf water isotopic composition matched observations, we would assume that plants were in steady state and that the isotopic composition of water transpired by the plants ($\delta^{18}O_t$, $\delta^D_t$) was equivalent to the isotopic composition of plant root water, which reflects source water. If leaf water departed from steady state predictions, we planned to model the isotopic composition of plant transpiration after Farquhar et al., (2005).

For the steady state test, we used the “string-of-lakes” model by Gat et al., (1991), which describes progressive enrichment along parallel veins of grasses in steady state as shown by Helliker et al., (2000). We then corrected for the Péclet effect using an expression of bulk leaf water as a mixture of enriched water in the leaf lamina and less enriched water in stems (Flanagan, 1993):

\[ \delta^{18}O_{\text{leaf}} = \delta^{18}O_{\text{model}} \times f + \delta^{18}O_{\text{stem}} \times (1 - f) \]  

(7)

where \( f \) is the proportion of the bulk leaf water subjected to mixing with stem water, the subscripts leaf and stem refer to bulk leaf water and stem water, respectively, and model refers to leaf water at the site of evaporative enrichment as calculated by the Gat & Bowser (1991) model.

To utilize source water measurements to determine the contribution of evaporation and transpiration fluxes to water vapor above the marsh, we assumed that there were three
primary sources of vapor within the marsh canopy: evaporation from standing or soil water, transpiration from *Typha*, and background vapor originating from outside the marsh. We also assumed that the relative contribution of the three sources did not significantly change during the measurement period, and that there was no condensation or other losses of water vapor other than by turbulent mixing with air. Typically, linear mixing models or “Keeling plots” relating the isotopic composition of water vapor to the inverse of water vapor concentrations are used to distinguish the isotopic composition of ET from that of the background vapor (Keeling, 1961, Moreira *et al.*, 1997, Yepez *et al.*, 2003, Williams *et al.*, 2004, Yepez *et al.*, 2005). We separately measured/modelled the isotopic composition of the “endmembers” or components of total canopy vapor: evaporation, transpiration, and background vapor. We used a mass balance approach based on Lai *et al.* (2006) to estimate the relative contributions of the sources to canopy vapor. The fractional contribution of each source to marsh canopy vapor is shown by

$$F_C = F_B + F_E + F_T,$$

where $F$ is the fractionation contribution and the subscripts C, B, E, and T stand for canopy vapor, background air, evaporation, and plant transpiration, respectively. Using conservation of mass,

$$\delta^{18}O_C F_C = \delta^{18}O_B F_B + \delta^{18}O_E F_E + \delta^{18}O_T F_T$$

and

$$\delta D_C F_C = \delta D_B F_B + \delta D_E F_E + \delta D_T F_T$$

The isotopic composition of background vapor ($\delta^{18}O_B, \delta D_B$) was assumed to be similar to vapor collected above-canopy (4 m above the marsh water/soil surface).

We solved for the fractional contributions of each source, and transpiration efficiency ($T/ET$) was calculated as a percent, $F_T/(F_E+F_T)*100$. We used the approach of Phillips *et al.*
To calculate the errors associated with this three-endmember source partitioning model, and took into account analytical error and correlation between $\delta^{18}O$ and $\delta D$.

### 4.3.7. Calculation of $T/ET_{EC}$

We used H$_2$O and CO$_2$ fluxes from two eddy covariance towers located in the SJFM to measure and partition ET. The two towers were located $<$500 m away from each other in ponds with similar vegetation, but different hydrologic regimes. The main tower (i.e. flooded pond) where isotopic measurements took place was flooded from January 5 to June 18, 2007, but the second tower (i.e. unflooded pond) was located in a nearby pond that was dry throughout 2007, which resulted in no growth of Typha leaf area. Further details on the two ponds can be found in Rocha & Goulden (2010).

Net ecosystem exchange (NEE) was partitioned into Gross Ecosystem Exchange (GEE) and Total Ecosystem Respiration (TER) using weekly light response curves as described by Rocha & Goulden (2010). T and GEE were then used to calculate an average Vegetation Water Use Efficiency (VWUE):

$$\text{VWUE} = \frac{T}{\text{GEE}}$$ (11)

$T/ET_{EC}$ was calculated using three methods that made different assumptions about how transpiration varied over the course of the season. The first method assumed that VWUE was constant and T was calculated by multiplying the seasonal average VWUE by GEE. For the second method, we obtained T for the period when standing water was present by multiplying the seasonal average VWUE by GEE. The evaporation of the unflooded pond
was similar to evaporation from the flooded pond following the disappearance of standing water, when plants were senesced (p=0.38). Thus, for the second method we assumed that evaporation from the unflooded pond ($E_S$) represented evaporation from the flooded pond in the period when standing water had disappeared and plants were alive. $E_S$ was subtracted from ET at the flooded pond during this period to obtain T. In the third method, T was extrapolated from the period when standing water was present by utilizing the symmetry of the seasonal T cycle. T estimates from the first method were extrapolated using a sinusoidal function with the day of year as an explanatory variable and assuming that maximum T occurred at the start of the dry period.

We limited our analysis to daily average midday values (time of day: 10:00 to 2:00) because they represented daily maximal rates under well mixed conditions, and used a 14 day moving average for smoothing. We incorporated uncertainty into the $T/ET_{EC}$ estimates using a bootstrapping approach (Zoubir, 1993), which applied a one standard deviation error for estimated VWUE and $E_S$. Average daily $T/ET_{EC}$ and 95% confidence intervals were calculated from 21 estimates of $T/ET_{EC}$ using the three methods and the bootstrapping approach. The goal of this exercise was to constrain $T/ET_{EC}$ estimates by incorporating multiple methodological ensembles that would yield a range of possible transpiration efficiencies.
4.4. Results

As there were linear relationships between $\delta^{18}$O and $\delta$D for all samples in this study (Figure 1a-c), we focus our results on $\delta^{18}$O to avoid redundancy.

![Figure 1 - The $\delta^{18}$O - $\delta$D relationships of (a) of within-canopy vapor collected 0.1 and 4 m above marsh standing water and modeled evaporation from marsh standing water and soil water (b) *Typha latifolia* L. root water and marsh standing water, (c) leaf waters. In all plots, the solid line indicates the California meteoric water line as determined by Kendall and Coplen (2001). Note that figures axes vary in scale.]

4.4.1. The isotopic composition of standing water, soil water, and root water

The isotopic composition of root water, standing water, and soil water were all evaporatively enriched relative to the Local Meteoric Water Line (LMWL) for California described by Kendall and Coplen (2001), as they fell to the right (Figure 1b). The marsh water height and sequence of events are shown in Figure 2a. Marsh standing water in January and February following initial flooding events was isotopically similar in 2007 and 2008 ($p>0.1$) (Figure 2). The isotopic composition of surface standing water was enriched following initial flooding on January 5, 2007 until February 22, 2007 ($p<0.01$), when it became depleted (Figure 2). From March 7, 2007 to June 15, 2007, there was an overall enrichment in the isotopic composition of standing water ($p<0.0001$). Standing
water disappeared on June 18, 2007 (Figure 2a). The isotopic composition of soil water was constant following June 28, 2007 (Figure 2b; p>0.1).

![Figure 2](image)

**Figure 2** – (a) Marsh water level during 2007-2008. (b) The isotopic composition of standing water, soil water, and *Typha latifolia* L. root water during 2007-2008. The asterisk (*) shows a marginal difference (p=0.5-0.1) between root water and standing water/soil water, and double asterisks (**) show a significant difference (p<0.05).

The isotopic composition of root water and standing water were similar, except on March 23, 2007, when standing water was enriched relative to root water (Figure 2b; p<0.05). Surface soil water (0-5 cm) was always isotopically enriched relative to root water, except on June 28, 2007 when root water was more enriched.
4.4.2. Leaf properties and isotopic composition

Over the *Typha* growing season in 2007, there was a decline in average daily leaf water content (Figure 3a; p=0.0005) and stomatal conductance (Figure 3b; p=0.0137). The average specific leaf area was 53.2 ± 7.1 (SD) cm²/g.

![Figure 3](image)

**Figure 3** – (a) Leaf water content over time for *Typha latifolia* L. (b) Stomatal conductance over time for *Typha latifolia* L.

The isotopic composition of leaf water was evaporatively enriched relative to the California LMWL (Figure 1c). Leaf water showed progressive isotopic enrichment from the base of the leaf to the tip of 18.6 ± 9.0‰ for δ¹⁸O (not shown; p<0.05). The average isotopic composition of whole leaves varied between 1.0 and 13.2 ‰ for δ¹⁸O, and became enriched over time on sampling days (p<0.0001) (Figure 4a). Modeling leaf water without taking into account the Péclet effect (Flanagan 1993) resulted in values that were isotopically enriched relative to observed leaf water by 3.5 ± 2.0 ‰ δ¹⁸O (Figure 4b). After taking into account the Péclet effect, the difference between measured and
modeled leaf water isotopic composition was $1.3 \pm 0.8 \% \delta^{18}O$ (Figure 4b). There was no trend in model residuals with time of day or over the growing season (p>0.1).

![Figure 4](image)

**Figure 4** – (a) Diurnal variation in isotopic composition of leaf water. (b) The isotopic composition of observed vs. modeled leaf waters, with and without the Péclet correction. The solid line is the 1:1 line.

### 4.4.3. The isotopic composition of water vapor

The isotopic composition of vapor samples collected within-canopy (0.1 m above the marsh water/soil surface) fell within $1.0 \pm 0.8 \% \delta^{18}O$ and $8.1 \pm 6.5 \% \delta D$ of the California LWML (Figure 1a). Vapor samples collected above-canopy (4 m above the marsh standing water/soil) fell within $5.2 \pm 6.5 \% \delta^{18}O$ and $0.7 \pm 0.8 \% \delta D$ (Figure 1a). A repeated measures ANOVA showed no overall difference between vapor above- and within-canopy (p>0.1) (Figure 5). However, there were differences between above- and within-canopy vapor during individual sampling events up to $5.3\% \delta^{18}O$ (July 13, 2007; Figure 6). The average isotopic composition of vapor derived from evaporated surface water was estimated to be $-31.2 \pm 7.8 \% \delta^{18}O$, and was the most
isotopically depleted possible source of local water vapor that we considered (Figure 5).

Evaporation was isotopically depleted relative to other days on January 16, 2008 as humidity was high (73%) and temperature was low (15ºC).

Figure 5 – The isotopic composition of endmembers used for distinguishing between evaporation and transpiration in canopy vapor (collected at 0.1 m height above marsh water/soil surface). The endmembers were background vapor collected 4 m above the marsh water/soil surface, evaporation from standing water and soil, and transpiration (assumed to be similar to *Typha latifolia* L. root water in 2007, and marsh standing water in 2008).
Figure 6 – Diurnal variations in the isotopic composition ($\delta^{18}$O and $\delta$D) of within-canopy vapor collected 0.1 and 4 m above marsh water or the soil surface on sampling days during the 2007 Typha latifolia L. growing season and in 2008 when standing water was present, but plants had greatly senesced. Note that figures axes vary in scale.

4.4.4. Estimates of T/ET

Total ET is shown in Figure 7a, and shows rates on average 2 mm/day during the spring and summer growing season, and <1 mm/day during winter when plants were senesced. T/ET$_{SW}$ estimates indicated that transpiration occurred at a higher rate than evaporation and accounted for most of the marsh water loss from March to June (Figure 7b; p<0.05). On June 15, 2007, three days prior to complete absence of standing water, there was only 6 cm of standing water left and 82 ± 2% of marsh ET loss since March 23, 2007 was due to transpiration (Figure 7b). Evaporation accounted for 3.6% of total water loss at the marsh, while drainage accounted for 81.4%.
Figure 7 – (a) Marsh ET, (b) T/ET<sub>SW</sub>, (c) T/ET<sub>EC</sub> and T/ET<sub>WV</sub> during study period.

T/ET<sub>WV</sub> ranged from 56 ± 17% to 96 ± 67% in the growing season of 2007 (Figure 7c). In 2008, when most plants had senesced and standing water was present, T/ET<sub>WV</sub> ranged from 38 ± 67% and 91 ± 30%. Background vapor comprised 83 ± 10% of canopy vapor for these measurements, and the average percent errors of T/ET<sub>WV</sub> estimates were 48% in
2007 and 82% in 2008. The highest percent errors (SD/T/ET) were 177.5% and 168.9% on February 13, 2009 and March 12, 2009, when the proportion of transpiration was low (T/ET is ≤ 40%) and the contribution of background vapor to the total water vapor concentration was high (>90%) (Figure 8a,b). Excluding these two data points, there was a linear relationship between the contribution of background vapor to the total vapor, and the percent error of the T/ET estimate (Figure 8b; p<0.01). Mean T/ET_{EC} estimated using eddy covariance measurements followed a seasonal cycle, being near zero when plants emerged, and increasing to ~85% during the peak growth period from June-August, and declining thereafter (Figure 7c).

![Figure 8](image)

**Figure 8** – The relationship between percent error (SD/T/ET) of T/ET_{WV} estimates and (a) T/ET_{WV} (%) and (b) % background water vapor in total canopy vapor.

### 4.5. Discussion

To provide context for interpreting the isotopic values of sources and fluxes used for constructing mass balances, we first discuss the isotopic variability within and among the liquid water pools. Leaf water was highly variable (7.6 ± 9.4‰ δ^{18}O) (Figure 4a), while
soil water and root water showed little variability (-1.1 ± 1.4‰ $\delta^{18}$O and -2.5 ± 0.8‰ $\delta^{18}$O, respectively) (Figure 2), comparable to Lai (2006) and Welp (2008). Standing water was initially 5.4‰ $\delta^{18}$O, and became progressively enriched by 3.8‰.

Canopy water vapor at 4 m was within 1.1 ± 1.0‰ $\delta^{18}$O of the isotopic composition of vapor collected simultaneously at a nearby beach (13.8 ± 1.0‰ $\delta^{18}$O) (p>0.1), suggesting a large influence of marine evaporation. The isotopic composition of vapor varied slightly (-13.6 ± 1.5‰ $\delta^{18}$O for within-canopy vapor and -14.1 ± 2.1‰ $\delta^{18}$O for above-canopy vapor). We observed diurnal variations as high as 2.3‰ $\delta^{18}$O within-canopy (January 16, 2008) and 9.1‰ $\delta^{18}$O above-canopy (July 13, 2007) in Figure 6, while Lai (2006) observed diurnal variations as high as 4‰ above and within an old-growth coniferous forest. Welp (2008) found the mean isotopic composition of vapor to vary over the season by nearly 15‰.

4.5.1. Evaporative influence on standing water

The isotopic composition of surface standing water was enriched following initial flooding at the marsh on January 5, 2007 (p<0.01), consistent with evaporation, until another pulse of water was added on February 21, 2007 (Figure 2). This resulted in isotopic depletion (Figure 2b). After flooding stopped on March 7, 2007, there was a steady evaporative enrichment in standing water until complete evaporation of surface water on June 18, 2007 (p<0.01). Evaporation was expected to cause isotopic enrichment of soil water following evaporation of standing surface water (i.e. Allison, 1982, Allison et al., 1983), but following June 28, 2007, the isotopic composition of soil water did not
change (p>0.1), possibly because of low evaporation rates from marsh soil. On June 7 and 28, 2007, the isotopic compositions of standing water and soil water, respectively, were anomalously low relative to the general trend of enrichment. On December 18, 2007, water was again diverted into the marsh for the next growing season (Figure 2a), and became evaporatively enriched following an additional pulse of water on January 19, 2008 (Figure 2b). Marsh standing water in January-February 2008 was depleted to values similar to the previous January-February period of 2007 due to re-flooding of unevaporated water. These values were within 0.39 ± 0.2 ‰ δ18O of the California LMWL and are consistent with an origin of primarily native recharge and local precipitation, rather than imported Colorado river water (Williams, 1997).

4.5.2. Péclet effect and steady state transpiration

The isotopic composition of leaf water was calculated using a steady state model and compared to observed leaf water. The results of these calculations, which predict the isotopic composition of water at the sites of evaporation, were enriched relative to observed leaf water by 3.5 ± 2.0 ‰ for δ18O (Figure 4b). Several studies have shown that Péclet corrections can improve the agreement between modeled and measured leaf water (Cernusak et al., 2002, Farquhar et al., 2003, Ripullone et al., 2008). To correct for the Péclet effect, we used 0.72 for f in Equation (7). This estimate fell within the range of other studies, which have reported that the pool of unfractionated water (1-f) varies between 0.10 to 0.30 of the total water volume (Allison et al., 1985, Leaney et al., 1985, Walker et al., 1989, Flanagan et al., 1991, Roden et al., 1999). As plant ET increases, 1-f is expected to increase in a fractional curvilinear fashion (Barbour et al., 2004). We did
not find any trend in the model residuals with time of day or over the growing season (not shown; p>0.1), implying that f remained constant. Consideration of the Péclet effect improved similarity between observed and modeled values (Figure 4b). Because Péclet-corrected steady state modeled values for δ¹⁸O of leaf water were similar to observed values (within 1.3 ± 0.8‰), we assumed that the plants operated primarily in isotopic steady state, concurrent with our hypothesis, and that the isotopic composition of transpired vapor was similar to the water found in plant roots. In winter 2008, when most plants had senesced and standing water was present, we assumed that the remaining green vegetation continued to operate in isotopic steady state and that the isotopic composition of transpiration was similar to that of marsh standing water. Transpiration was thus the most enriched source of canopy vapor (Figure 5). During the daytime, short crop canopies similar to our 1 m tall canopy have been shown to be at or near to steady state (Yakir & Wang 1996, Welp et al., 2008).

4.5.3. Marsh water balance and comparison of T/ETₘₖ, T/ETₖₚ and T/ETₑₖₚ

T/ETₘₖ showed a progressively larger proportion of total marsh water lost via transpiration over the course of the season (Figure 7b). This indicates a higher proportion of transpiration to evaporation, as we hypothesized and as suggested by Goulden et al., (2007), who found low rates of subcanopy water flux (evaporation) and high rates of water flux during the growing season due to cool water temperatures in the same marsh. Our findings agree with other studies that have suggested that wetland transpiration exceeds open water evaporation (Price 1994; Herbst & Kappen 1999; Pauliukonis & Schneider 2001; Acreman et al. 2003). T/ETₘₖ represents the cumulative rather than the
instantaneous transpiration efficiency, and although not directly comparable to T/ETEC, both methods agreed that transpiration exceeded evaporation for most of the growing season (Figure 7b,c). On June 15, 2007, when only 6 cm of standing water remained, 82 ± 2% of total, cumulative ET loss since March 23, 2007 could be attributed to transpiration (Figure 7b). Only a small amount of total marsh water loss was due to evaporation (3.6%), while transpiration accounted for 15.0%, and the largest loss was from subsurface drainage (81.4%). The large drainage loss was unexpected as others have suggested that ET represents that greatest water loss from wetlands, and that the inflow of groundwater is more important than outflow (Owen, 1995, Hunt et al., 1996, Drexler et al., 2004).

The local influence of ET has been shown to be detectable in atmospheric vapor in several studies (e.g. Brunel et al., 1992, Brunel et al., 1997, Moreira et al., 1997, Yepez et al., 2003). In our study, the isotopic mass balance of canopy vapor showed that there was an influence of ET in canopy vapor during some sampling events (Figure 7c). ET was not detectable when the isotopic composition of above-canopy background vapor at 4 m and canopy vapor at 0.1 m were similar, which suggests strong turbulent mixing. On these days, the isotopic composition of atmospheric water vapor was sometimes outside of the range of possible sources (evaporation, transpiration, and background vapor), indicating that strong mixing may have transported water vapor sources from outside of the marsh area. The isotopic composition of canopy vapor was within the range of sources when vapor above- and within-canopy were isotopically distinct, and when the water vapor content in the canopy exceeded the content above-canopy, indicating the
presence of a local ET signal in the canopy due to stable atmospheric conditions. Specifically, this occurred when the difference between above- and within-canopy vapor exceeded $0.19\%\delta^{18}O$ and $4.0\%\delta D$ in 2007, and $0.8\%\delta^{18}O$ and $0.8\%\delta D$ in 2008, and when the canopy atmospheric water vapor content exceeded above-canopy content by 0.9 g H$_2$O/m$^3$ in 2007, and 0.25 g H$_2$O/m$^3$ in 2008. These conditions occurred only on 38% of sampling days. The amount of ET in canopy vapor varied considerably, from $8 \pm 5\%$ on May 4, 2008 to $42 \pm 12\%$ on January 18, 2008. Overall, the local contribution of ET was small, as the average was $17 \pm 10\%$ of canopy vapor.

$T/ET_{WV}$ ranged from $56 \pm 17\%$ to $96 \pm 67\%$ with an average of $78 \pm 20\%$ in the growing season of 2007 (Figure 7c). This is within the range reported by Herbst et al., 1999, who found $T/ET$ up to 83%. Mean values of $T/ET_{EC}$ was very similar to $T/ET_{WV}$ during the growing period, and following evaporation of standing water (Figure 7c). At least 39% or more of ET in the canopy can be attributed to transpiration according to $T/ET_{WV}$, but the large error limits the usefulness of these calculations and their ability to detect temporal patterns. We expected temporal patterns as seen in $T/ET_{EC}$, which increased as the growing season progressed and waned as *Typha* senesced.

In winter 2008, when most plants had senesced and standing water was present, $T/ET_{WV}$ ranged from $38 \pm 67\%$ to $91 \pm 30\%$, or $55 \pm 26\%$ on average (Figure 7c). Although the errors of $T/ET$ estimates were high, they often did not overlap with zero (Figure 7c), indicating an influence of transpiration. It was somewhat surprising that transpiration was detected during this period due to relatively low green leaf area. However, because the
evaporation rate was also likely low due to cooler water and higher relative humidity in this winter period (Goulden et al. 2007), a detectable fraction of T/ET is plausible. Alternatively, water vapor from transpiration of other vegetation, such as riparian forests approximately 40 m away may have been present in these samples, and during other periods in the study. Uncertainties in the measurement footprint complicate the interpretation of studies that rely on concentration rather than direct flux measurements.

The large errors in the T/ET\textsubscript{WV} estimates are likely due to a relatively small contribution of ET to total atmospheric water vapor. The two estimates with the highest percent error (SD/T/ET\textsubscript{WV}), 177.5% and 168.9% are during the presence of standing water and senescence of Typha in February and March 2008. These occurred when the proportion of transpiration was low (T/ET\textsubscript{WV} is ≤ 40%), and the contribution of background water vapor was very high (>90%) (Figure 8a,b). Excluding these two extreme data points, the percent error of the T/ET\textsubscript{WV} estimate increased with proportion of background vapor in the total canopy water vapor (p<.01) (Figure 8b). This relationship can provide a guideline for conditions that are favorable to distinguishing between evaporation and transpiration; i.e., when the contribution of non-local, background water vapor to the total is less than about 80%.

4.5.4. Plant water use

Surface standing water was sometimes isotopically enriched relative to root water, indicating that evaporative enrichment primarily affected the surface of the standing water pool, and that roots were accessing deeper, less depleted water (Figure 2b). Root
water was initially similar in isotopic composition to marsh surface water when plants were very young on March 4, 2007 (p>0.1), but as plants grew, the isotopic composition of root water became depleted relative to surface water (p<0.05 on March 23, 2007), suggesting use of deeper water. Roots were isotopically depleted relative to surface soil water in August and October, following the disappearance of standing water, which suggests transpiration of deeper soil water (Figure 2b). Deep rooting was suggested by Goulden (2007), who found that high rates of ET continued following the disappearance of standing water. In this study, declines in leaf water content and stomatal conductance indicated that plants experienced increased water stress over the course of the season (Figure 3; p<0.05), but sustained high rates of T/ET (Figure 7b,c), probably due to deep roots.

4.6. Conclusions

The δ¹⁸O and δD isotopic composition of water pools (leaf, root, standing, and soil water) and fluxes (transpiration, evaporation) showed that: 1) plants were generally transpiring in isotopic steady state without large short-term changes in leaf water content. However, there were longer-term, seasonal declines in leaf water content concurrent with declines in stomatal conductance. 2) The isotope mass balance of standing marsh water was more useful in distinguishing among pathways of water loss than the isotope mass balance of water vapor. 3) There were differences in isotopic composition and water vapor content above- versus within-canopy during some sampling periods, which suggested an influence of local sources of transpiration and evaporation in the canopy, but other periods indicated strong atmospheric mixing. This was due to the many periods in which
atmospheric mixing obscured the influence of local sources on atmospheric water vapor. Under more stable atmospheric conditions, the water vapor approach yielded improved results. Yet both the isotope mass balance and eddy covariance measurements suggested a large influence of transpiration on total ET, despite the presence of standing water. In general, measurements of the isotopic composition of liquid water pools in standing water, soil water, and plants provided very useful information about local ecohydrology and water balance, while distinguishing between evaporation and transpiration with the isotopic composition of water vapor was possible only under specific atmospheric conditions, such as when there was a large vertical gradient in water vapor concentrations and isotopes.

4.7. Acknowledgements

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Chapter 5

Water sources of urban trees in the Los Angeles metropolitan area

5.1. Abstract

In semi-arid cities, urban trees are often irrigated, but may also utilize natural water sources such as groundwater. Consequently, the sources of water for urban tree transpiration may be uncertain, complicating efforts to efficiently manage water resources. We used stable isotopes to determine tree water sources in the Los Angeles basin, where we hypothesized that trees would rely on irrigation water in the soil rather than develop deep roots to tap into groundwater. We evaluated the oxygen ($\delta^{18}O$) and hydrogen ($\deltaD$) isotope ratios of xylem water, irrigation water, soil water, and groundwater in a study of temporal patterns in water sources at two urban sites, and a study of spatial patterns at nine urban sites and one “natural” riparian forest. We found that groundwater consisted of local precipitation and recharge from irrigation. Contrary to our hypothesis, we found that despite frequent irrigation, some trees tap into groundwater, although in most species this was a small water source. Some trees appeared to be using very shallow soil water at <30 cm depth, suggesting that these mature urban trees were quite shallowly rooted. In the natural site, trees appeared to be using urban runoff in addition to shallow soil water. We were able to identify tree uptake
of precipitation at only 3 sites. The results of this research show that some irrigated trees utilize groundwater and do not rely solely on irrigation water, which may make them able to withstand drought and/or water conservation measures. However, some irrigated trees may develop very shallow root systems, which may make them more susceptible. In addition, reduced irrigation may decrease groundwater recharge which could impact groundwater height and tree uptake of groundwater.

5.2. Introduction

Trees are widely planted in urban areas for many reasons, including atmospheric cooling, carbon sequestration, and air pollution removal (Akbari, 2002, McPherson et al., 2005, Mueller et al., 2005, Nowak et al., 2006, Nowak et al., 2007), as well as aesthetics. In arid and semi-arid areas such as southern California, trees are usually irrigated and therefore utilize water resources, which already may be scarce. It is important to understand sources of water to urban trees to assist landscapers and city planners in managing these resources. If trees primarily utilize irrigation water for transpiration, they may be susceptible to reductions in irrigation in response to water conservation measures. If trees rely heavily on rainwater, they may be susceptible to drought. On the other hand, if urban trees rely primarily on groundwater, changes in irrigation management may have fewer impacts on urban tree performance. While 14-30% of municipal water consumption is used outdoors for irrigation in California (Gleick, 2003), only a few studies have explored the water use of urban trees (Bush et al., 2008, Wang et al., 2008, McCarthy et al., 2010, Pataki et al., in press), and we are not aware of previous studies that identified specific sources of transpiration.
Plant access to groundwater depends on rooting depth, which can be positively correlated with aboveground plant size (Canadell et al., 1996, Schenk et al., 2005). In unirrigated natural ecosystems and plantations, plant type, climate, and soil variables can explain as much as 50% of rooting depth, which increases with aridity (Schenk et al., 2002). In urban areas, rooting depth may depend on both available soil volume and irrigation frequency. In well-watered soils, urban trees may be expected to develop shallow roots in order to capture irrigation inputs. In addition, urban tree rooting depth may be limited by restricted soil volumes or compacted soils (Gilman et al., 1987, Day et al., 1994, Grabosky et al., 2004). Soils near asphalt, concrete, and other structures may physically limit root growth. In addition, soils in urban areas can be both unintentionally and intentionally compacted to sustain the weight of pavement or other structures, which may inhibit root penetration. Thus, urban tree rooting depth may depend on species, management practices, or the environmental and physical variables at a particular site.

Stable isotopes of oxygen and hydrogen (δ¹⁸O and δD) are commonly used to study tree water sources and rooting depths in a variety of ecosystems (Dawson et al., 2002). Because there is assumed to be no isotopic fractionation during uptake of soil water by plant roots, stem water represents a mixture of the isotopic composition of plant water sources. By understanding the isotopic composition of possible sources or “endmembers,” it is possible to identify and partition the sources using measurements of the isotopic composition of stem water, provided that sources are isotopically distinct.
In this study, we measured the isotopic composition of possible tree water sources at several sites in the Los Angeles basin. These included irrigation water, rain water, and groundwater (tap water was also measured for comparison with irrigation water). We also measured both the temporal and spatial variability of soil water and stem water of urban trees from several horticultural settings. These measurements were used to address the questions: (1) What is the isotopic composition of possible source waters to urban trees, and how do they vary temporally and spatially? (2) Is soil water comprised of evaporated irrigation water? (3) If groundwater is isotopically distinct from soil water, what is the proportion of groundwater use relative to soil water use, and how does this vary spatially, and among species? (4) Do tree water sources change over time, particularly in response to precipitation events? We hypothesized that the isotopic composition of irrigation would change very little over time, and that soil water would consist of evaporated irrigation water. We also hypothesized that trees would rely on soil water rather than groundwater due to frequent irrigation which would likely result in shallow rooting depths. Finally, we expected that the isotopic composition of tree stem water would not change over time, except in response to large precipitation events. The results of this study can inform urban forest planning and management as well as a general understanding of the ecohydrology of irrigated, semi-arid cities.

5.3. Methods

5.3.1. Study sites

This study was conducted at multiple sites throughout the Los Angeles (LA) Basin in California, USA. The Basin is a sediment-filled coastal plain bounded by mountain
ranges. It has a Mediterranean climate, with rainfall occurring primarily in the winter and spring months. During the study period (2005-2008), the average annual temperature was 16.8 ± 0.13°C and precipitation was 27.9 ± 17 cm (www.cimis.water.ca.gov, Irvine station #75). The Basin is highly urbanized, with a population over 16,000,000 in the Los Angeles Riverside-Orange County consolidated metropolitan area, according to the 2000 U.S. census.

The study sites measured in this study are summarized in Table 1. Several have been studied previously by McCarthy and Pataki (2010) and Pataki et al. (in press), which measured tree water use with constant heat sap flux sensors. In the present study, we examined water sources of the same trees measured in the previous studies. Therefore, for some species and sites, information about plant water use and water relations is available. For each species at a particular site, we selected 5 individuals per species for isotopic analysis, except where noted.
**Table 1** - Characteristics of nine urban sites and one natural site, an unmanaged riparian forest, measured in this study.

<table>
<thead>
<tr>
<th>Site name</th>
<th>Year sampled</th>
<th>Urban/natural</th>
<th>Irrigation</th>
<th>Species studied</th>
<th>Coordinates</th>
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<tbody>
<tr>
<td>UCI 2007</td>
<td>2007</td>
<td>Urban</td>
<td>Yes</td>
<td>Canary Island pine <em>(Pinus canariensis C. Sm)</em> California sycamore <em>(Platanus racemosa Nutt.)</em></td>
<td>33.65 N, 117.85 W</td>
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<td>Street Trees</td>
<td>2007</td>
<td>Urban</td>
<td>No</td>
<td>London planetree <em>(Platanus hybrida Brot.)</em> California sycamore</td>
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<td>2007</td>
<td>Urban</td>
<td>Yes</td>
<td>silk floss <em>(Chorisia speciosa A. St.-Hil.)</em> jacaranda <em>(Jacaranda mimosifolia D. Don)</em></td>
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<td>LA Zoo Non-irrigated</td>
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<td>Urban</td>
<td>No</td>
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<tr>
<td>Natural</td>
<td>2007</td>
<td>Natural</td>
<td>No</td>
<td>London planetree coastal live oak <em>(Quercus agrifolia Née)</em></td>
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<tr>
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<td>redwood <em>(Sequoia sempervirens (Lamb. ex D. Don) Endl.)</em></td>
<td>33.65 N, 117.85 W</td>
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<td>LA Police Academy</td>
<td>2008</td>
<td>Urban</td>
<td>Yes</td>
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<td>Yes</td>
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<td>Urban</td>
<td>Yes</td>
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<td>34.15 N, 118.05 W</td>
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</table>
Our study consisted of two parts: The first was repeated sampling of two groups of trees on the campus of University of California (UC), Irvine to assess temporal patterns of water isotopes in 2005-2006. The second consisted of sampling at multiple sites throughout the LA Basin in 2007-2008 to assess spatial variability. We sampled 5 sites per year in 2007 and 2008 (10 total sites), which spanned a large coastal-inland gradient and differed in management.

In the first study of temporal dynamics, the two study sites were designated “UCI-1” and “UCI-2.” At UCI-1, we studied red river gum (*Eucalyptus camaldulensis* Dehnh.), stone pine (*Pinus pinea* L.), Canary Island pine (*Pinus canariensis* C. Sm.), New Zealand Christmas tree (*Metrosideros excelsa*), and flax leaf paperbark (*Melaleuca linariifolia* Sm.). At UCI-2, we studied Norfolk Island pine (*Araucaria heterophylla* (Salisb.) Franco), stone pine (*Pinus pinea* L.), and jacaranda (*Jacaranda mimosifolia* D. Don). The trees were planted on an understory of primarily turfgrass, with some ice plant (*Carpobrotus chilensis* (Molina) N.E. Br.) and ivy (*Hedera helix* L.). At UCI-1 and UCI-2, we sampled 3 individuals per species.

In 2007, we measured five sites including a site at the UC Irvine campus, where we measured Canary Island pine (*Pinus canariensis* C. Sm.) and California sycamore (*Platanus racemosa* Nutt.). At this site, which we refer to as “UCI 2007,” trees were planted on an understory of turfgrass, ice plant, and ivy. This site is referred to as the “Irrigated” site by *McCarthy and Pataki* (2010) and “Campus” by *Pataki et al.* (in press). Another site consisted of street trees growing in a narrow strip of soil along a sidewalk in
Los Angeles, referred to as “Street Trees” by this study, *McCarthy and Pataki* (2010), and *Pataki et al.* (in press). At this site, we measured three individuals of California sycamore and four London planetree (*Platanus hybrida* Brot.). Although these species are related and may hybridize (*Rhymer et al.*, 1996), they were treated separately as their sap flux rates differed (*Pataki et al.*, in press). These trees were not directly irrigated or fertilized, although they likely received irrigation runoff from adjacent lawns and gardens. Two other sites were located adjacent to each other at the Los Angeles Zoo and Botanical Garden. We measured Canary Island pine and laurel sumac (*Malosma laurina* (Nutt.) Nutt. ex Abrams) in an unirrigated site with an herbaceous understory, called “LA Zoo Non-Irrigated.” We measured jacaranda (*Jacaranda mimosifolia* D. don) and silk floss (*Chorisia speciosa* A. St.-Hil.) in the adjacent site which did receive irrigation, called “LA Zoo Irrigated.” These sites are considered together by *Pataki et al.* (in press) and are called “Los Angeles Zoo.” We studied one non-urban site to serve as a comparison to the urban sites. This site, called “Natural,” was located in a riparian forest in Starr Ranch Sanctuary, a 4000-acre National Audubon Society preserve located in the foothills of the Santa Ana Mountains, with urban commercial and residential development on its western border. Trees at this site were naturally established (not planted), and received no irrigation, fertilization, or other management, in contrast to the urban sites.

In 2008, we measured another set of five sites for additional spatial sampling. One site was located at the Los Angeles Police Academy, where a mature urban forest and rock garden has been managed as a public garden since 1935. Measurements at this “LA
Police Academy” site were conducted on Canary Island pine, Chinese elm (*Ulmus parvifolia* Jacq.), and redwood (*Sequoia sempervirens* (Lamb. ex D. Don) Endl.). A variety of shrubs and herbaceous plants were found in the understory. Two additional sites were located at the Los Angeles County Arboretum and Botanic Garden in Arcadia, California. This botanical garden and urban park is jointly managed by the Los Angeles Arboretum Foundation and the Los Angeles County Department of Parks and Recreation. We called a section of this garden where trees originated from Australia, “LA Arboretum Site A” and a section where trees originated from South America, “LA Arboretum Site SA,” as by Pataki et al. (in press). Although Site SA was located in the South American section, it contained some species from other regions. At Site A, we measured lacebark (*Brachychiton discolor* F.J. Muell) and kurrajong (*Brachychiton populneus* (Schott & Endl.) R. Br.). At Site SA, we measured crape myrtle (*Lagerstroemia indica* L.), goldenrain (*Koelreuteria paniculata* Laxm.), laurel fig (*Ficus microcarpa* L. f.), jacaranda (*Jacaranda mimosifolia* D. Don.), and honeylocust (*Gleditsia triacanthos* L.). These sites did not contain an understory, and there were several trees of similar or other species present in the plots. At UC Irvine, we selected another site, “UCI 2008,” where we focused our measurements on a stand of redwood trees. There was no understory present at this site although there were individuals of California sycamore and eucalyptus present nearby. We conducted additional measurements of redwoods at Fullerton Arboretum at California State University.

Additional information about sites, species, and their water-use can be found in papers by McCarthy and Pataki (2010) and Pataki et al. (in press).
5.3.2. *Stem, soil, irrigation, and runoff sampling*

At each sampling event, we collected a small, woody non-green stem (xylem) sample about 5 cm long and 3-9 mm in diameter, from 3-5 individuals of each species. Stems were quickly shaved to eliminate the possibility of evaporatively enriched water near the bark. Soil cores were sampled with a 5 cm corer (AMS Inc., American Falls, Idaho). Irrigation samples were collected directly from sprinklers found at sites. At the Natural site, we also collected water from puddles of runoff found at the site during sampling events. Stem, soil, irrigation, and runoff samples were immediately placed in airtight vials, sealed with parafilm, and stored frozen. Prior to isotopic analysis, water was extracted from stems and soils using cryogenic vacuum distillation (West *et al.*, 2006).

In the temporal study on the UC Irvine campus in 2005-2006, stems were collected once during fall, winter and spring. Soils were collected closely following stem sampling events. In the fall, soils were collected at 3-6 cm, 17-20 cm, and 29-32 cm (n=3) and in winter and spring at 17-20 cm (n=5). Irrigation samples were collected from sprinklers at each site on 28 November 2005 and weekly or fortnightly following February 2006. Tap water was also taken on a weekly basis for comparison with irrigation water.

In the spatial study during 2007-2008, soils were sampled in three locations per site, and were taken from 0-10 cm and 20-30 cm depth, except at the two LA Zoo sites where soils were sampled from 0-10 cm and 10-20 cm, and the Fullerton Arboretum, where soils were sampled at 0-10 cm, 10-20 cm, and 20-30 cm. Irrigation samples from sprinklers installed at all sites were collected 1 or 2 times during the collection period for the stems.
For the sites sampled in 2007, we collected stems from each species in August, and again on 27 September 2007, five days after a 10.66 mm rainfall event (www.cimis.ca.gov, Irvine station #75). Soils at each site were collected approximately three weeks later, in October. For sites sampled in 2008, we collected stems from each species 2 or 3 times during a period from August and October. Soils were collected in October, on the last collection day for stems.

5.3.3. Rain and groundwater sampling

Rainwater was collected from each rainfall event from 1 December 2005 to 22 April 2006 at the UC Irvine campus. Samples were funneled into screw cap vials during the rainfall event, immediately sealed with parafilm, and stored frozen.

Rainwater was collected during the 20-22 September 2007 rainfall event in mineral oil-capped beakers which were inserted in the soil at the UCI 2007, Street Trees, Irrigated LA Zoo, Non-irrigated LA Zoo, and Natural sites. No later than one day following rainfall, rainwater samples were decanted to remove oil, stored in screw cap vials, sealed with parafilm, and stored frozen.

Groundwater sampling was conducted at two canal locations in Orange County (33.69 N, 117.82 W and 33.71 N, 117.80 W) in November of 2007 and 2008. At these times, canal water levels were very low, and exposed canal walls were leaking groundwater through small orifices or “weepholes.” We withdrew water from these weepholes using a syringe, and after discarding the first 5 withdrawals, we injected the water into vials. Groundwater
was similarly collected from weepholes along the Los Angeles river (34.06 N, 118.23 W). Groundwater from the Natural site was sampled from a large on-site groundwater well on 2 November 2007. The well was actively pumped as it was a source of tap water at the site. All groundwater samples were collected in screw cap vials, sealed with parafilm, and frozen until analysis.

5.3.4. Isotope analysis

All water samples were analyzed for oxygen and hydrogen isotope ratios (δ¹⁸O and δD) using a thermal conversion elemental analyzer (TC EA, Thermofinnigan, San Jose, CA) coupled to an Isotope Ratio Mass Spectrometer (Delta Plus XP, Thermofinnigan, San Jose, CA) following Gehre et al., 2004. Oxygen and hydrogen isotope measurements were expressed in common δ notation. δ¹⁸O and δD was referenced to V-SMOW with a precision of 0.23‰ and 1.1‰ (SD), respectively.

5.3.5. Statistical analyses

Statistical analyses were performed on SAS 9.1.3 software (SAS Institute Inc., Cary, NC). Repeated measures analyses of variance (ANOVA) was conducted for time-series of stem water isotope values using the General Linear Model for Analysis of Variance. Post-hoc tests were conducted using the Tukey Standardized Range Test. Two-sided t-tests were performed using Excel 2003 (Microsoft Corp., Redmond, WA) to compare between the isotopic compositions of species and soil depths at a particular site. When there were more than two species at a site, analyses of variance (ANOVA) were performed to compare stem water isotope values of various species using the General
Linear Model for Analysis of Variance. For all statistical analyses, \( p < 0.05 \) was considered significant, and \( p < 0.1 \) was considered marginally significant.

5.3.6. Determination of water sources

We constructed least squares linear regressions of the isotopic composition of soil water (evaporation lines) measured at 0-30 cm to determine whether the source of soil water was primarily evaporated irrigation water (Gat et al., 1991b, Gat, 1996). The initial isotopic composition of unevaporated water, evaporated water, and the remaining enriched fraction form “evaporation lines” (Gat, 1996) that have been used previously to determine water sources (e.g., Ortega-Guerrero et al., 1997, Corbin et al., 2005).

Water in the unsaturated zone of the soil may become isotopically enriched relative to groundwater due to evaporative losses to the atmosphere (Busch et al., 1992, Brunel et al., 1995). Evaporative enrichment in \( \delta^{18}O \) and \( \delta D \) of water typically decreases with depth from the unsaturated soils at the surface to the saturated zone below where liquid transport dominates (Allison 1982, Allison et al., 1983). We assumed that the soil evaporation line we constructed using measurements in the top 0-30 cm was representative of the entire soil profile, although we did not measure soil deeper than 30 cm. To establish the soil water source, we determined whether the isotopic composition of irrigation fell within the standard error (SE) of the soil evaporation line. For example, Figure 3(a) shows a soil evaporation line and irrigation water value for the UCI-1 site on 25 October 2005. Since the isotopic composition of irrigation water falls within the
standard error of the soil evaporation line, we would conclude that soil water is comprised of evaporated irrigation water in this case.

To calculate tree water sources, stem water was considered to be a mixture of two possible endmembers: groundwater and soil water. To determine the proportion of transpiration from groundwater, we determined the linear equation of the mixing line between groundwater and soil water, represented by the distance between soil and groundwater, or $D_{s-g}$ as shown for the LA Police Academy site in Figure 4(a). We then calculated the intersection of the mixing line and the soil evaporation line and assumed that this point reflects the average isotopic value of soil water taken up in transpiration (Figure 4a). This value was used to calculate the proportion of groundwater present in stem water using the equation from Thorburn and Walker (1993), which is the ratio of the distance between stem water and soil water ($D_{s-t}$) to the distance between groundwater and soil water ($D_{s-g}$).
5.4. Results

5.4.1. Irrigation, groundwater, and rainwater

The isotopic composition of irrigation samples fell within 0.47 ± 0.38‰ (SD) for δ^{18}O and 3.6 ± 2.9‰ for δD of the California meteoric water line (MWL) reported by Kendall et al., (2001) (Figure 1). Irrigation water measured at the LA Police Academy deviated from this line more than other sites, and fell to the right by 1.1 ± 0.6‰ for δ^{18}O and 8.8 ± 4.8‰ for δD.

![Figure 1 - The isotope ratios (δ^{18}O and δD) of all irrigation, groundwater, soil water, stem water, and rainwater samples taken in this study. A linear regression through the rainwater isotopes (local meteoric water line) is shown with SE. The California meteoric water line by Kendall & Coplen (2001) is also shown.](image-url)
Irrigation water at UC Irvine was initially isotopically similar to tapwater, but increased by ~2‰ between March 2006 and July 2006 (Figure 2a, p<0.0001) and ~20‰ for δD (not shown, p<0.0001). Tapwater declined ~2‰ between June 2005 and January 2006 (Figure 2a, p<0.0001) and ~24‰ for δD (not shown, p<0.0001). The isotopic enrichment in δ¹⁸O of irrigation waters varied spatially as follows: LA Arboretum > UCI 2007, LA Zoo, UCI 2008, Fullerton Arboretum > Street Trees > LA Police Academy (Figure 2b, p<0.05). The isotopic composition of irrigation at the LA Arboretum was enriched relative to the LA Police Academy by 3.5 ± 0.08‰ for δ¹⁸O (Figure 2b) and 39.7 ± 2.4‰ for δD (not shown). However, the isotopic composition of irrigation at the LA Arboretum was similar to groundwater (Figure 2b). At the other study sites, δ¹⁸O of irrigation water was isotopically depleted relative to groundwater (Figure 2a, b).

**Figure 2** - (a) Time-series of δ¹⁸O of rain, irrigation, and tap water at UC Irvine in 2005-2006. (b) Average δ¹⁸O of rain and irrigation water with SE collected at sites throughout the Los Angeles Basin in 2007-2008. Groundwater collected from urban areas on two occasions in November of 2007 and 2008 is shown as a bar with SE.
Groundwater samples fell within 0.27 ± 0.23 in δ18O and 2.1 ± 1.8 in δD of the statewide meteoric water line (Figure 1). Groundwaters collected at the urban sites in 2007 and 2008 did not differ from one another in δ18O or δD (Table 2, p > 0.1). Groundwaters at the Natural site were slightly enriched relative to groundwaters collected at urban sites by 0.9 ± 0.2 ‰ δ18O and 11.5 ± 2.1‰ δD (Table 2, p < 0.0001).

Table 2 - δ18O and δD of groundwater with SD.

<table>
<thead>
<tr>
<th>Location</th>
<th>Year collected</th>
<th>δ18O</th>
<th>δD</th>
</tr>
</thead>
<tbody>
<tr>
<td>Urban sites</td>
<td>2007</td>
<td>-6.8 ± 0.2</td>
<td>-47.6 ± 1.5</td>
</tr>
<tr>
<td>Urban sites</td>
<td>2008</td>
<td>-6.9 ± 0.2</td>
<td>-46.6 ± 2.1</td>
</tr>
<tr>
<td>Natural site</td>
<td>2008</td>
<td>-5.9 ± 0.2</td>
<td>-35.5 ± 0.7</td>
</tr>
</tbody>
</table>

A linear regression through rainwater samples formed a local meteoric water line (LMWL) for our study region and fell to the left of the statewide MWL by 1.0 ± 0.5 δ18O and 8.2 ± 4.1 δD (Figure 1). δ18O of rain water was highly variable both temporally (Figure 2a) and spatially (Figure 2b), but was almost always enriched relative to groundwater and irrigation water. The pattern of δD was similar (not shown). There was no strong seasonal trend in the isotopic composition of UC Irvine rain water, but there was temporal variability in rain events of up to 9.0‰ for δ18O and 53.7‰ for δD. In the study of spatial variability, the largest isotopic difference among sites was the enrichment of LA Zoo rainwater relative to UCI 2007 rainwater during the 20-22 September 2007 rain event by 4.4 ± 0.6‰ for δ18O and 33.0 ± 3.5‰ for δD (Figure 2b).
5.4.2. The isotopic composition of soil water

All soil water isotopes in this study fell to the right of the LMWL, indicating evaporatively enriched water (Figure 1). The isotopic composition of soil water formed linear relationships between $\delta^{18}O$ and $\delta D$ (soil evaporation lines) at each site except UCI-1 on 10 February 2006 and LA Zoo Irrigated (Figures 3-6) ($p<0.05$). At UCI 2008 and Street Trees, the isotopic composition of shallow soil water (0-10 cm) was enriched relative to deeper soil (10-20 cm or 20-30 cm) ($p<0.05$) (Figures 5b,c). At UCI 2007 and the Natural site, the isotopic composition of shallow soil water was marginally enriched relative to deeper soil ($p<0.1$) (Figures 5a,e). At LA Zoo Non-irrigated, $\delta D$ of shallow soil water was significantly enriched relative to deeper soil ($p<0.05$), and $\delta^{18}O$ of shallow soil water was marginally enriched ($p<0.1$) (Figure 5d). At other sites, there were no significant differences in the isotopic composition of soil water between depths.

In the temporal study at UCI-1 and UCI-2 in 2005-2006, ANCOVAs of $\delta D$ with $\delta^{18}O$ as the covariate tested for differences in the intercepts of soil evaporation lines during the different seasons. The results showed that soil water at both sites became enriched as a result of spring rainfall (Figures 3c,f). Winter UCI-1 10 February 2006 soil water was excluded from the analysis due to lack of linearity.

The isotopic composition of irrigation water was within the SE of the soil evaporation lines at the UCI-1 and UCI-2 on all dates (Figure 3), and at the UCI 2007 and UCI 2008 sites (Figure 5a,b). However, this was not the case at the other irrigated sites (Figures 4, 5c-e). At the two non-irrigated urban sites, Street Trees and LA Zoo Non-irrigated, the
isotopic composition of irrigation from adjacent locations did not fall within the SE of the soil evaporation lines (Figure 5c,d). The isotopic composition of runoff did not fall within the SE of the soil evaporation lines at the Natural site (Figure 5e).

Figure 3 - Average $\delta^{18}$O and $\delta$D with SE of stem waters and soil waters collected at UC Irvine sites UCI-1 (a-c) and UCI-2 (d-f) in 2005-2006. The sampling on 17 April 2006 occurred 3 days following a large rainfall event (13.5 mm, www.cimis.water.ca.gov, Irvine station #75). Linear regressions of soil water isotopes (soil evaporation lines) are shown with SE, with n=3 at each soil depth in October 2005, and n=5 at the 17-20 cm soil depth in February 2006 and April 2006. Groundwater collected from urban areas is shown with SE. Irrigation water collected on site is shown with SE.
Figure 4 - Average $\delta^{18}O$ and $\delta D$ with SE of stem water at the LA Police Academy, Fullerton Arboretum, LA Arboretum Site A and SA sites. Groundwater and irrigation averages of $\delta^{18}O$ and $\delta D$ with SE are also depicted. Linear regressions of soil water isotopes (soil evaporation lines) are shown with SE (n=3 at each soil depth). The distance between soil and groundwater is shown by a dotted line ($D_{s-g}$), and the distance between soil and stem water is a gray line ($D_{s-t}$). The intersection point of the soil evaporation line and $D_{s-g}$ is indicated by a star. Note figures differ in scale.
Figure 5 - Average $\delta^{18}O$ and $\delta D$ with SE of stem water at the UCI 2007, UCI 2008, Street Trees, LA Zoo Non-irrigated, and Natural sites. Groundwater and irrigation averages of $\delta^{18}O$ and $\delta D$ with SE are also depicted. At the non-irrigated sites (c-d), the isotopic composition of irrigation water from an adjacent location is shown. At the Natural site (e), the isotopic composition of runoff water found at the site is shown. Linear regressions of soil water isotopes (soil evaporation lines) are shown with SE (n=3 at each soil depth). Note figures differ in scale.
Figure 6 - Isotope values of stem waters at the LA Zoo Irrigated site before and after rainfall events on 20-22 September 2007 totaling 10.66 mm. The rainfall events enriched stem water isotopic composition of jacaranda and silk floss ($p<0.05$). Arrows show the direction of movement in the stem isotopes following rainfall. Groundwater and irrigation averages of $\delta^{18}O$ and $\delta D$ with SE are also depicted. Soil water isotopes are shown. Soil waters at this site did not form a linear relationship.
5.4.3. The isotopic composition of stem water

There was no change in the isotopic composition of stem waters collected during a rainless 2-3 month period in late summer/fall for any species or site in 2008 (Table 3) (p>0.1).

Table 3 - $\delta^{18}$O of stem water for sites sampled in 2008 with SE. There was no isotopic difference between stem waters of species collected in different months (p>0.05).

<table>
<thead>
<tr>
<th>Site</th>
<th>Species</th>
<th>August</th>
<th>September</th>
<th>October</th>
</tr>
</thead>
<tbody>
<tr>
<td>UCI 2008</td>
<td>redwood</td>
<td>-6.6 ± 1.5</td>
<td>-5.8 ± 0.3</td>
<td></td>
</tr>
<tr>
<td>LA Police Academy</td>
<td>Chinese elm</td>
<td>-7.2 ± 0.4</td>
<td>-6.9 ± 0.3</td>
<td>-8.2 ± 0.3</td>
</tr>
<tr>
<td></td>
<td>Canary Island pine</td>
<td>-6.4 ± 1.4</td>
<td>-7.8 ± 0.4</td>
<td>-5.0 ± 1.0</td>
</tr>
<tr>
<td></td>
<td>redwood</td>
<td>-5.8 ± 1.0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fullerton Arboretum</td>
<td>redwood</td>
<td>-5.9 ± 0.4</td>
<td>-5.6 ± 0.6</td>
<td>-5.3 ± 1.0</td>
</tr>
<tr>
<td>LA Arboretum Site A</td>
<td>jacaranda</td>
<td>-5.1 ± 0.3</td>
<td>-4.9 ± 0.5</td>
<td></td>
</tr>
<tr>
<td></td>
<td>crape myrtle</td>
<td>-5.2 ± 0.3</td>
<td>-4.7 ± 0.4</td>
<td></td>
</tr>
<tr>
<td></td>
<td>goldenrain</td>
<td>-5.4 ± 0.5</td>
<td>-5.0 ± 1.4</td>
<td></td>
</tr>
<tr>
<td></td>
<td>honeylocust</td>
<td>-5.2 ± 0.4</td>
<td>-4.8 ± 0.6</td>
<td></td>
</tr>
<tr>
<td></td>
<td>laurel fig</td>
<td>-5.6 ± 0.6</td>
<td>-5.6 ± 0.1</td>
<td></td>
</tr>
<tr>
<td>LA Arboretum Site A</td>
<td>kurrajong</td>
<td>-5.1 ± 0.1</td>
<td>-5.6 ± 0.1</td>
<td></td>
</tr>
<tr>
<td></td>
<td>lacebark</td>
<td>-5.3 ± 0.4</td>
<td>-5.7 ± 0.6</td>
<td></td>
</tr>
</tbody>
</table>

Thus, stem waters collected during these months were averaged for each species in Figures 4 and 5. There were species differences in stem water isotope ratios at some sites. At the UCI-1 site, flax leaf paperbark was enriched in $\delta D$ above Canary Island pine and New Zealand Christmas tree in fall (October 2005) (p=0.0234) (Figure 3a), marginally above all other species in winter (February 2006) (p=0.0649) (Figure 3b), and above New Zealand Christmas tree and red river gum in spring (April 2006) (p=0.0138) (Figure 3c). There were no differences in $\delta^{18}$O of stem waters among species at the UCI-1 site. At the UCI-2 site, there were no differences in isotopic composition among species in fall (October 2005) and winter (February 2006) (Figures 3a,b), although stem water of
jacaranda became more enriched than other species following rainfall (Figure 3c). At the LA Police Academy, stem water of redwoods was isotopically enriched in δ¹⁸O and δD relative to Chinese elm and Canary Island pine (p<0.05), which did not differ from each other (Figure 4a). In the Natural site, California sycamore was enriched relative to coastal live oak in δ¹⁸O (p = 0.0335) and δD (p = 0.0394) (Figure 5e). At the remaining sites (Figures 4b,c,d and Figures 5a,b,c,d), there were no differences in the isotopic composition of stem waters among species.

Stem water isotope ratios of several species appeared to be affected by rain events. Three days following a large rainfall event (13.5 mm, www.cimis.water.ca.gov, Irvine station #75) on 17 April 2006, all species at the UCI-1 site became enriched in δ¹⁸O (p=0.0029) and δD (p=0.0004) (Figure 4). There were no large rainfall events (> 6.4 mm) for more than three weeks prior to the other sampling times. The isotopic composition of irrigation did not change during the stem sampling period (Figure 3a) (p>0.05). At the UCI-2 site, stem waters in jacaranda became isotopically enriched relative to other species following spring rainfall in δ¹⁸O (p=0.0129) and marginally in δD (p=0.0800), while stem water of the other species did not change (p>0.05) (Figure 5).

Isotopic enrichment following rainfall was also observed in two irrigated species, silk floss and jacaranda, at the LA Zoo Irrigated site, five days following rainfall events totaling 10.66 mm from 20-22 September 2007 (Figure 6). Silk floss became enriched in δ¹⁸O (p=0.02) and jacaranda became enriched in both δ¹⁸O (p=0.03) and δD (p=0.009). The isotopic composition of stem water at the other sites (UCI 2007, Street Trees, LA
Zoo Non-irrigated, Natural) did not follow this pattern of enrichment in δD or δ\textsuperscript{18}O (not shown). Thus, stem waters collected before and after rainfall were averaged for species at these sites in Figure 5. Notably, stem water isotopic composition at the Non-irrigated LA Zoo site, which was directly adjacent to the Irrigated LA Zoo site, did not become enriched following rainfall.

5.4.4. Tree water sources

We estimated tree water sources with two-ended or three-ended mixing models, depending on the site and the possible number of water sources. The resulting estimated proportions of groundwater use relative to soil water are shown in Table 4. Most of these calculations used a two-ended mixing model, but at the Natural site, there were three possible water sources (groundwater, runoff, and soil water). Therefore we used the approach of Phillips et al., (2001) and determined that 24 ± 11% and 10 ± 30% of tree water use was from groundwater in California sycamore and coastal live oak, respectively (Table 4).
Table 4 - Percentages of groundwater use relative to soil water use for species with SE. The proportion of groundwater in stem water was calculated based on Thorburn and Walker (1993). At the Natural site, a three-endmember mixing model between groundwater, soil water (0-30 cm depth), and runoff water at the site was used to calculate the proportion of groundwater use.

<table>
<thead>
<tr>
<th>Site</th>
<th>Species</th>
<th>Groundwater use (%) (Avg ± SE)</th>
</tr>
</thead>
<tbody>
<tr>
<td>LA Police Academy</td>
<td>Chinese elm</td>
<td>4 ± 5 %</td>
</tr>
<tr>
<td></td>
<td>Canary Island pine</td>
<td>24 ± 4 %</td>
</tr>
<tr>
<td></td>
<td>redwood</td>
<td>18 ± 6%</td>
</tr>
<tr>
<td>Fullerton Arboretum</td>
<td>redwood</td>
<td>0 ± 7 %</td>
</tr>
<tr>
<td>LA Arboretum Site SA</td>
<td>jacaranda</td>
<td>32 ± 14 %</td>
</tr>
<tr>
<td></td>
<td>crape myrtle</td>
<td>0 ± 14 %</td>
</tr>
<tr>
<td></td>
<td>honeylocust</td>
<td>0 ± 9 %</td>
</tr>
<tr>
<td></td>
<td>goldenrain</td>
<td>5 ± 25 %</td>
</tr>
<tr>
<td></td>
<td>laurel fig</td>
<td>55 ± 18 %</td>
</tr>
<tr>
<td>LA Arboretum Site A</td>
<td>lacebark</td>
<td>84 ± 9 %</td>
</tr>
<tr>
<td></td>
<td>kurrajong</td>
<td>90 ± 10%</td>
</tr>
<tr>
<td>Natural</td>
<td>California sycamore</td>
<td>24 ± 11 %</td>
</tr>
<tr>
<td></td>
<td>coastal live oak</td>
<td>10 ± 30 %</td>
</tr>
</tbody>
</table>

Groundwater use was as high as 90% in species at the LA Arboretum Site A site. It was not possible to estimate water sources at the UCI 2008 and Street Trees sites because groundwater and soil water were not isotopically distinct, that is, groundwater fell within the SE of the soil evaporation line (Figures 5b,c). At UCI-1 and UCI-2 (Figure 3), UCI 2007 (Figure 5a), Street Trees and LA Zoo Non-irrigated (Figure 5d), and the Natural site (Figure 5e), stem water fell to the right of both groundwater and the soil evaporation line, indicating that species at these sites were either using an additional source of water which was not measured, or that stem water was isotopically enriched due to stem evaporation or mixing with leaf water.
5.5. Discussion

5.5.1. The isotopic composition of source waters

Irrigation values fell close (within 0.47 ± 0.38 δ¹⁸O and 3.6 ± 2.9 δD, Figure 1) to the California MWL reported by Kendall and Coplen (2001), which was constructed using samples of California river waters, because irrigation is primarily derived from the same snowmelt waters which flow into California rivers. The similarity of measured irrigation to the MWL indicates relatively little evaporation from these sources. Deviations from the California MWL (Figure 1) may occur because water imports are used to supplement local sources of groundwater during portions of the year (DWR, 2003). Generally isotope values of low-latitude, low-elevation coastal regions are enriched relative to inland, high-latitude, and mountainous areas (Bowen et al. 2007). The isotope values of irrigation water were depleted relative to coastal low-elevation precipitation collected in this study, indicating that irrigation was primarily derived from inland, mountainous snowmelt. The isotope values of groundwater were generally in between isotope values of local precipitation and irrigation water (Figure 1), indicating an influence of both sources and the importance of irrigation in recharging groundwater. In the case of the Los Angeles Arboretum, the isotopic composition of irrigation water was similar to groundwater for δ¹⁸O (Figure 2b) and δD (not shown).

In general, plant water sources were isotopically distinct at most sites, although there was temporal variability. We found changes in the isotopic composition of irrigation water over time, in contrast with our hypothesis (Figure 2a). We are not aware of other studies that have explored long-term trends in the isotopic composition of irrigation waters. At
UC Irvine, the source of irrigation water is recycled wastewater that has undergone tertiary treatment. Therefore, the difference between the isotopic composition of irrigation water and tap water at UC Irvine (Figure 2a) may be due to time lags in the reclamation process, and to evaporative enrichment during treatment. The trends in the isotopic composition of irrigation may also reveal variations in urban water imports or management of tap water. A previous study of sources to Orange County aquifers showed that water imported from the Colorado River aqueduct is highly depleted relative to native Santa Ana river recharge and local rainfall (Williams, 1997). Thus, declines or increases in the isotopic value of tap water may represent increasing or decreasing influence of imported water from the Colorado River and other snowmelt sources. The average isotopic composition of tapwater at UC Irvine (-9.9±0.5 δ18O and -76.9±5.4 δD) was similar to that reported by Bowen et al., (2007) for Fullerton, California (-9.7±0.2 δ18O and -77±2 δD).

The proximity of groundwater samples to the MWL reported by Kendall and Coplen (2001) (0.27 ± 0.23 ‰ for δ18O and 2.1 ± 1.8 ‰ for δD), indicates that these waters were not subject to evaporation, and are consistent with high-elevation snowmelt as shown by Williams (1997). Groundwater collected in the urban area was similar over time from 2007-2008 (Table 2, p>0.05), and the spatial variability was very small: only 0.2‰ for δ18O and 2.0‰ for δD (SD). However, Natural site groundwaters were slightly enriched relative to groundwaters collected in the urban areas by 0.9 ± 0.2 ‰ δ18O and 11.5 ± 2.1‰ δD (Table 2, p<0.0001). Because the urban areas are highly irrigated, they may
have some infiltration of irrigation to groundwater, resulting in isotopic depletion relative to natural sites.

The isotopic composition of rainwater was highly variable both temporally (Figure 2a) and spatially (Figure 2b) as reported previously (Dansgaard, 1964, Clark et al., 1997). Data from southern California have shown typical inverse correlations of $\delta D$ and $\delta^{18}O$ with increasing latitude, altitude, and distance from coast (Williams et al., 1997). The isotopic composition of rainwater was typically enriched relative to groundwater, irrigation, and tap water (Figure 2 a,b). Because tap water is derived from high-elevation snowmelt water sources, it is reasonable to expect that it would be isotopically depleted relative to local precipitation (Bowen et al., 2007).

The isotopic composition of soil water is generally considered to be influenced by rainwater which has subsequently experienced isotopic enrichment by evaporation (Tang et al., 2001). In this study of urban soils, irrigation waters were also a large input of water to surface soils. Although we were not able to quantify the volume of irrigation input into the soil, the study region has a Mediterranean climate with little summer precipitation, and irrigation inputs were the primary water source during this period. The isotopic composition of soil water formed linear relationships between $\delta^{18}O$ and $\delta D$ at all sites except UCI-1 on 10 February 2006 and LA Zoo Irrigated (Figures 3b, 6). At the UCI-1 and UCI-2 sites (Figure 3) and the UCI 2007 and UCI 2008 sites (Figures 5a,b), our results were consistent with evaporated irrigation water as the primary water source. However, at the other irrigated urban sites (Figure 4a-d), the soil evaporation lines were
depleted relative to irrigation water. This indicates that there is a source of water to these soils that was not measured. One possibility is urban runoff water; the isotopic composition of runoff (-11.1 $\delta^{18}$O, -94.1‰ $\delta$D) collected in a storm drain near the LA Police Academy was more negative than irrigation (-10.7 $\delta^{18}$O, -86.7‰ $\delta$D). Runoff water may be derived from irrigation waters from other sites, tap water, or wastewater.

At the non-irrigated sites (Street Trees, LA Zoo Non-Irrigated, and Natural), the soil evaporation line fell above the value for irrigation water from nearby locations or runoff (Figures 5c-e), suggesting that the source of water to the soil was more enriched than irrigation water. This is likely because rainwater, which tends to be enriched relative to irrigation water, plays a more important role in non-irrigated soils.

5.5.2. Tree water sources

Several assumptions were made for the calculations of tree groundwater use in this study. It is known that surface soils can become isotopically enriched due to evaporation, and that a linear relationship can be formed between $\delta^{18}$O and $\delta$D of evaporated soil water (Allison et al., 1983, Brunel et al., 1995). We measured the isotopic composition of surface soil (<30 cm depth), and assumed that if there was soil water below 30 cm subject to evaporation, then the $\delta^{18}$O and $\delta$D would follow the same linear relationship. We also assumed that the primary sources of stem water were evaporatively enriched soil water and groundwater, and that the isotopic enrichment found in stem water was solely due to uptake soil water, rather than stem evaporation or mixing with leaf water. At some sites this assumption was violated, and tree water source attribution was not possible. At UCI-
1 and UCI-2 (Figure 3), UCI 2007 (Figure 5a), Street Trees (Figure 5c) and LA Zoo Non-
irrigated (Figure 5d), and the Natural site (Figure 5e), stem waters fell to the right of both
groundwater and the soil evaporation line, indicating that species at these sites were
either using an additional source of water which was not measured, or that stem water
was isotopically enriched due to stem evaporation or mixing with leaf water.

This study suggests that urban trees, at least in some locations, tap into groundwater in
addition to soil water derived from evaporated irrigation and other surface water inputs
such as rainwater (Table 4), contrary to our hypothesis. As much as 90% of tree water use
was derived from groundwater. This will clearly vary spatially depending on rooting and
water table depth. However, trees at all locations in our study appeared to utilize
shallow, evaporated soil water to at least some degree, with some species supplementing
shallow water use with unevaporated groundwater, and some relying almost exclusively
on soil water. Groundwater use constituted a small proportion of total water use at the
Fullerton, Police Academy and Natural sites (Table 4). Fullerton and the LA Police
Academy were irrigated, so low groundwater use may be explained by high moisture
availability at the soil surface. However, the unmanaged Natural site was unirrigated, so a
lower proportion of groundwater use in the fall was unexpected, as surface soil moisture
was very low during this period (McCarthy et al., 2010). However, McCarthy and Pataki
(2010) showed reductions in transpiration rates at this riparian site in the fall, indicative
of water stress. In addition, this site appeared to be influenced by urban runoff present in
small pools at the soil surface, which was isotopically depleted relative to both measured
groundwater and soil water (Figure 5e). The result of a three-endmember mixing model
(Phillips et al., 2001) showed that 56 ± 8% and 53 ± 20% of tree water use was from surface runoff in California sycamore and coastal live oak, respectively. Runoff may have been a source of water for non-irrigated urban trees as well; at the non-irrigated urban sites, the Street Trees and Non-Irrigated LA Zoo site, trees appeared to be using a source of water which was not measured in this study, as the isotopic composition of stem water fell to the right of groundwater and the soil evaporation lines (Figure 5c,d).

The highest groundwater use was at the LA Arboretum Site A site (84-90%) (Table 4). Trees at this site were large and may have been deeply rooted, as rooting depth can be related to aboveground plant size (Canadell et al., 1996, Schenk et al., 2005). This site was also watered less frequently than the other urban sites (unpublished data).

We determined that when the intersection point of the groundwater-soilwater mixing line and the soil evaporation line was within the range of the isotopic composition of surface soil water (<30 cm), then trees were using some proportion of soil water <30 cm. This was the case for the species at the LA Police Academy, Fullerton Arboretum, LA Arboretum Site A (Figure 4a-c), and the laurel fig and jacaranda at Arboretum site SA (Figure 4d). Thus, these mature urban trees appeared to be using very shallow soil water (<30 cm). However, the intersection points were depleted relative to soil water at <30 cm for the crape myrtle, goldenrain, and honeylocust at the LA Arboretum Site SA, suggesting that all water uptake occurred below 30 cm (Figure 4d). These trees appeared to be using a combination of groundwater and soil below 30 cm depth. At the UCI 2007, Street Trees, LA Zoo Non-Irrigated, and Natural sites, it was not possible to construct mixing lines between groundwater and soil water for stem waters, but the isotopic
composition of stem waters were depleted relative to shallow soil water (<30 cm), suggesting that trees were using at least some proportion of water deeper than 30 cm (Figures 5a,c,d,e).

There were differences in species water sources within sites (Table 4). At the Natural site, California sycamore used a greater proportion of groundwater than coastal live oak (24 ± 11 % vs. 10 ± 30 %). At the LA Police Academy site, groundwater use varied such that Canary Island pine > Chinese elm > redwood. Groundwater use may have been related to tree size, the Canary Island pines at this site were very large, with diameter at breast height (DBH) of 61.6 ± 4.6 cm, compared to 39.4 ± 6.8 cm for redwood and 28.9 ± 2.5 cm for Chinese elm. At the LA Arboretum Site SA, crape myrtle, honeylocust, and goldenrain used little or no groundwater, whereas jacaranda and laurel fig used >30% groundwater. This difference was not explainable by DBH, geographic origin, or climate in area of origin.

The enrichment of some stem waters following rainfall suggests that a few trees utilized rain water, in support of our hypothesis, though this was not the case for most species or sites. At the UCI-1 site, all species appeared to be using rainwater (Figure 3a-c). At the UCI-2 site, there were no differences in isotopic composition among species in fall (October 2005) and winter (February 2006), implying similar patterns of water uptake, and only jacaranda showed evidence of rainwater uptake following rainfall (Figure 3d-f). At the LA Zoo Irrigated site, two out of four species became isotopically enriched as a result of rainfall: jacaranda and silk floss (Figure 6). Interestingly, we did not find
significant use of rainwater at the adjacent Non-Irrigated LA Zoo site, probably because trees there were more deeply rooted. Jacaranda at two sites (Irrigated LA Zoo and UCI-2) appeared to be using rainwater, suggesting that this species may opportunistically take up rainwater with shallow roots.

5.6. Conclusions & Implications

Based on the uptake of very shallow soil water (<30 cm) by some mature urban trees in this study, our research reinforces the suggestion that irrigated trees can develop shallow root systems, which may make them more susceptible to drought or reduced irrigation. This study also supports the hypothesis that some urban trees tap into groundwater to supplement water received by irrigation, although this is very site- and species-specific. This has important implications for management of both trees and groundwater: at some sites supplemental irrigation may not be needed, or may be needed in much smaller quantities than assumed. However, as groundwater supply may be dependent on recharge from irrigation, trees that utilize groundwater may also require sustained irrigation. Conversely, uptake of groundwater may not be desirable where water tables are closely managed as a water source. Finally, these results also highlight the uncertainties in the urban water budget, which can be constrained with measurements of stable isotopes. At some irrigated sites, there appear to be unknown source(s) of water which are isotopically depleted relative to irrigation water. Similarly, tree water use at some sites was not explained solely by the isotopic composition of groundwater and irrigation water. Trees at these sites may be exploiting different sources of water, perhaps from runoff, storm drains, leaky pipes, or water sources at intermediate soil depths, which requires further
exploration. Stable isotope measurements at these sites show that the water budgets of these urban forests are not fully understood, and that a more complete accounting of the water isotope mass balance can improve our understanding of the ecohydrology of urban landscapes.

5.7. Acknowledgements

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Chapter 6

Conclusions

6.1. General conclusions

This dissertation has provided experimental evidence in southern California that N₂O fluxes can serve as a positive feedback to global warming in lawns, due to a positive response of N₂O fluxes to warming. Soil moisture is also positively correlated with N₂O fluxes, and irrigation as well as fertilization leads to higher N₂O fluxes. In addition, warming may exacerbate weed invasions, which may require more intensive management, e.g. herbicide application, to manage species composition. With regard to lawn water use, I define the irrigation efficiency of lawns as the amount of applied water that is allocated to plant transpiration. In a controlled study of the complete water budget of three experimental lawns, I found that irrigation efficiency varied from 16-43%, depending on management practices. The highest irrigation rate and lowest irrigation efficiency was in the landscape with a conventional irrigation system on a timer. The lowest irrigation rate and highest irrigation efficiency was in a landscape that utilized a soil moisture sensor-based irrigation system. At least half of the water applied to these landscapes was drained below the rooting zone. In addition, the “reference ET” method of calculating ET of lawns from meteorological measurements greatly over-estimated actual ET. Recommended rates of water application based on reference ET calculations may lead to over-irrigation. My results show that there is potential for climate change
mitigation in lawn management with practices that reduce N₂O emissions, and for climate change adaptation with practices that may reduce lawn water use in response to predicted future water scarcity and drought.

I utilized the stable isotope mass balance of water sources to provide information about urban water budgets and ecohydrological processes. Specifically, I determined water budget components of a local marsh, and water sources of trees in the Los Angeles basin. Stable isotope techniques applied at the marsh provided great insight about its water budget and function, providing support for primary water loss by transpiration and deep rooting depth of cattail plants. I discovered that the mass balance approach of standing water was more robust than the mass balance of water vapor, due to a large proportion of marine background vapor in air that often obscured the local water signal. Using stable isotopes, I also found that despite frequent irrigation, some trees tap into groundwater, although this was usually a small water source. The study showed that irrigated, mature urban trees may develop shallow roots due to uptake of shallow soil water (<30 cm). Some trees at a natural site appeared to be using urban runoff as a water source.

6.2. Summary of Results

In Chapter 2, I showed that N₂O fluxes increased in response to temperature and fertilization, in support of my hypothesis. Soils in the heated plots were enriched in δ¹⁵N relative to control plots, which suggested greater gaseous losses of nitrogen. I also found that C₄ crabgrass with relatively low nitrogen content appeared to be using nitrogen from mineralized organic matter, as the C/N ratio of C₄ plants was negatively correlated with
plant $\delta^{15}N$. This might be an effective strategy to obtain nitrogen when supply is low, and may facilitate competition with C$_3$ plants. Surprisingly, the $\delta^{13}C$ of bulk-harvested grasses indicated that the crabgrass weed was 30 percent more prevalent in the heated plots than in the unheated controls after less than a year of treatment. While laboratory studies have shown that warming favors C$_4$ photosynthesis, I did not expect such rapid confirmation in the field. This study suggested that N$_2$O emissions could be a positive feedback from climate change, which may also change grass community composition. The N$_2$O emissions were also strongly related to soil moisture, which highlights the importance of lawn irrigation as a control on both nitrogen and water losses.

I conducted a complete accounting of the water budgets of three landscapes in Chapter 3, which ranged in decreasing management intensity. As I expected, the amount of irrigation applied was highest in the Typical landscape, in which irrigation was controlled with an automatic timer. However, it was surprising that the Low-impact landscape with a weather-based irrigation system received only slightly less water. The lowest irrigation rate was in the Retrofitted landscape, which was fitted with a soil-moisture based irrigation system. There was no significant difference in soil moisture among landscapes, except in spring when the Retrofitted landscape was marginally drier than the Low-impact landscape. The warm-season paspalum in the Retrofitted landscape was expected to transpire less than the cool-season grasses in the other landscapes, but average annual ET was similar. The stomatal conductance of native sedge in the Low-impact landscape was lower than the other two species, and but this was offset by increased leaf area. Within-canopy vapor pressure deficit was very low (<0.6 kPa), suggesting low
evaporative losses. The Typical landscape had the lowest irrigation efficiency (ET/applied irrigation) of 16%, and the Low-impact landscape had only a slightly higher IE of 24%, while the highest IE of 43% was of the Retrofitted landscape. Runoff was less than 2% of applied irrigation. This study showed that the soil-moisture based sensor may be more sensitive to plant available water than the weather based system, and that excess irrigation water does not improve turfgrass function, but instead promotes drainage.

The low vapor pressure deficit within the lawn canopy (<0.6 kPa) indicated that ET saturated the non-turbulent air space and that the lawn had a significant boundary layer resistance. This means the evapotranspiration from the lawn was primarily in equilibrium with available net incoming radiation, and mostly independent of stomatal conductance and saturation deficit of the air. Thus the transfer of water vapor out of the canopy was primarily reliant on conductance from the canopy boundary layer. This suggests a possible limitation to the chamber approach of measuring ET. The chamber blocks the ion

The isotopic composition of leaf water variance can be directly related to wind speed (Daudet et al., 1999). Thus, the lack of wind in the chamber may cause underestimation of ET. I plan to validate my ET measurements by estimating ET using multiple methodologies. The final paper will include a comparison of ET measurements using various approaches.

In Chapter 4, I developed and tested two stable isotope methods to help quantify water budget components, and compared them to estimates from an eddy-covariance based technique which also separated evaporation from transpiration. The isotopic composition
of modeled leaf water was similar to observed leaf water, indicating that plants were transpiring in isotopic steady state and that the isotopic composition of transpiration was equivalent to the plant source water (root water). The marsh water became progressively enriched over time due to evaporation, while the other of components of water loss did not alter its isotopic composition. This enabled me to determine that transpiration accounted for most (82 ± 2%) of the cumulative ET from March to June. Evaporation accounted for 3.6% of the total water loss, transpiration accounted for 15%, and subsurface drainage accounted for the majority, 81.4%. I also evaluated the isotopic mass balance of marsh water vapor, and found that the proportion of transpiration to ET was high, but that the error was large, and the method tended to overestimate transpiration during periods when transpiration was low. The greatest uncertainty was associated with periods when atmospheric water vapor was dominated by non-local, background sources (>90%). Independent estimates of T/ET using eddy covariance measurements yielded similar high values of mean T/ET during the Typha growing period. The isotopic mass balance of standing water was more useful than the mass balance of water vapor as it provided more definitive results.

I continued my investigation of water losses using stable isotopes in urban systems in Chapter 5, this time with a focus on urban trees. Contrary to my hypothesis, I found that despite frequent irrigation, some trees tap into groundwater, although this was usually a small water source. Some mature trees appeared to be relying upon very shallow roots, based on their observed use of shallow soil water (<30 cm). In the one natural site I studied, trees appeared to be using urban runoff in addition to shallow soil water.
Precipitation was not an important source of water to trees, as only three urban sites showed uptake of precipitation immediately following a rain event. The water sources of trees in this study varied greatly in both sites and species. Because some irrigated trees utilized groundwater in addition to irrigation water, they may be able to withstand reductions in surface inputs caused by water conservation or other measures that reduce irrigation. However, some trees developed very shallow root systems, suggesting reliance on irrigation and the possibility of water stress in response to declines in irrigation.

6.3. Implications for future work

I found a positive response of N₂O emissions to warming in Chapter 2. A next step could be to determine the mechanism driving this response. Fertilization increases the nitrogen substrates available for both microbial processes of nitrification and denitrification, which are responsible for N₂O flux. Future studies could assess the extent to which these processes are influenced by warming. This could also be pursued using stable isotopes, as nitrification produces N₂O with a greater abundance of $^{15}$N in the terminal position of the molecule (Perez et al., 2006, Sutka et al., 2006). In addition, I found significant emissions of N₂O in turfgrass, and extrapolating these emissions based on the surface area of California (13,890 km²) would increase the 1999 estimate of N₂O emissions from agricultural soils in California by 14% (Franco, 2002, Milesi et al., 2005). However, to appropriately scale these estimates to whole regions, N₂O measurements should be taken in a variety of turfgrass species, soil types, ages, and climates. My study also found a significant increase in the community composition of C₄ vs. C₃ plants in response to warming, which likely plays a role in altering water and nitrogen cycles in turfgrass
and/or other systems, and should be evaluated with further studies. I also found evidence for mineralized organic matter utilization of C$_4$ plants, which may represent a mechanism for obtaining nutrients under nitrogen stress. Studies evaluating rooting characteristics of C$_4$ plants could help identify sources of mineralized organic matter as a competitive strategy for C$_4$ plants.

The study of landscape water budgets in Chapter 3 showed that drainage resulted in a large amount of water loss from urban landscapes, while runoff was negligible. It will be important to confirm this finding in soil types with more clay, which may have greater runoff losses relative to drainage. The direct chamber-based ET measurements of turfgrass in this study were about 80% lower than estimates from the California Irrigation Management Information System (CIMIS). This suggests that recommended watering rates for turfgrass should be adjusted, which will require measurement of ET from a variety of species, management types, and soil types, and subsequent modeling with environmental parameters. The observation that ET was not different among species in Chapter 3 suggests that modeling efforts will be more dependent on environmental conditions than species or turfgrass type. The low irrigation efficiencies of these landscapes highlight the need for optimizing irrigation efficiency in turfgrass. It would be worthwhile to determine the level of irrigation which supports the minimum ET loss necessary to maintain turf quality.

The stable isotope mass balance techniques I tested in Chapter 4 may be applied to other areas of southern California to determine urban water budget components, including
separate estimates of evaporation and transpiration. The standing water technique may be applied in other wetlands, and the water vapor technique may be applied in areas of the Los Angeles basin. However, my results showed that the mass balance of vapor must be applied with caution, as periods with high background vapor are prone to large error. As southern California is a coastal area usually subject to offshore wind, there may often be a large proportion of marine evaporation in water vapor. In addition, future studies may attempt to investigate the footprint of vapor collection, to understand which sources may be spatially integrated in a sample.

My study of urban tree water sources in Chapter 5 indicated that species were primarily using irrigation water, with some species supplementing irrigation with groundwater. However, there were clearly some trees which appeared to be using unknown sources of water. A natural progression of this line of research would be to determine these sources, which may be water from runoff, storm drains, leaky pipes, or water sources at intermediate soil depths. This study highlighted that the water budgets of urban forests are not fully understood, and that a more complete accounting of the isotopic mass balance could improve understanding of urban ecohydrology.

**Irrigation drainage implications to regional water budget**

This study detected high levels of drainage from irrigated urban landscapes. This suggests the importance of irrigation to local groundwater recharge. In fact, groundwater isotope values were in between isotope values of local precipitation and irrigation water, indicating the importance of both sources in recharging groundwater (Chapter 5, Figure
1). Recharge by irrigation can enhance the local groundwater supply, which is used to supplement tap water sources (DWR 2003). However, there are a number of reasons why recharge of groundwater by irrigation could cause environmental concern. Irrigation drainage can has been demonstrated to carry agricultural pollutants to groundwater in California (Charbonneau et al., 1993). In addition, the energy cost of importing water sources for irrigation is high. It has been estimated that the energy for moving and heating water is a fifth of energy use in California (Cooley, 2007). Although the LA Aqueduct transports water from the Owens Valley to Los Angeles by gravity, the second Los Angeles Aqueduct, Colorado Aqueduct, California Aqueduct require supplemental energy to transport water. Finally, water exports cause declines in water levels of Mono Lake, which increases air pollution as well as reduces migratory bird habitat (Reisner 1993).

Trees which utilize groundwater may be reliant upon recharge by irrigation. Recharge by irrigation could raise the groundwater table, making groundwater accessible to tree roots. Thus, if irrigation is reduced, groundwater tables may be lowered, reducing water availability for trees. As irrigation supplied by nonlocal sources is directly or indirectly (by groundwater recharge) a source of plant transpiration in semiarid urban areas such as Los Angeles, it could contribute to an increase in urban humidity (Carreiro et al., 2008). This could alter precipitation patterns, as well as contribute to local or global warming.

Irrigation drainage has implications for groundwater recharge, urban tree water use, and possibly climate, which should be investigated. Future work will expand the findings of
this dissertation to regional or statewide scales. Isotopic mass balances as well as flux measurements will add to knowledge about the components of the urban water budget and complex interrelationships with environmental variables, and how water balance may respond to global changes.
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