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Characterizing Water and Nitrogen Dynamics in Urban/Suburban Landscapes

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CHARACTERIZING WATER AND NITROGEN DYNAMICS IN URBAN/SUBURBAN LANDSCAPES

by

Hongyan Sun

A dissertation submitted in partial fulfillment of the requirements for the degree of DOCTOR OF PHILOSOPHY in Plant Science

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2011
ABSTRACT

Characterizing Water and Nitrogen Dynamics in Urban/Suburban Landscapes

by

Hongyan Sun, Doctor of Philosophy
Utah State University, 2011

Major Professor: Dr. Kelly Kopp
Department: Plants, Soils, and Climate

This research investigated the water use of different plant types in urban landscapes, nitrogen (N) and water transport in turf, and potential N leaching from urban landscapes to ground water. In the first study, three landscape treatments integrating different types of plants—woody, herbaceous perennial, turf—and putative water use classifications—Mesic, Mixed, Xeric—were grown in large drainage lysimeters. Each landscape plot was divided into woody, turf, and herbaceous perennial plant hydrozones and irrigated for optimum water status over two years, with water use measured using a water balance approach. For woody plants and herbaceous perennials, canopy cover, rather than plant type or water use classification, was the key determinant of water use relative to reference evapotranspiration (ETo) under well-watered conditions. For turf, monthly evapotranspiration (ETa) followed a trend linearly related to ETo. In the second study, water transport parameters were calibrated using an inverse simulation with Kentucky bluegrass (KBG). Subsequently, those parameters were applied to simulate water use by tall fescue (TF) and buffalograss (BG) turfgrasses using numerical modeling (Hydrus-1D). By using the calibrated soil hydraulic parameters obtained from the
water transport simulation, N transport and transformation was modeled with Hydrus-1D under different irrigation rates and different fertilization rates. Different soil texture scenarios were also simulated to demonstrate the influence of soil texture on N leaching. In the third study, the simulated N-leaching from different soil textures was integrated into a Geographic Information System (GIS) approach to estimate \( \text{NO}_3^-\)N leaching mass from urban turf areas. Nitrate-N leaching risks to ground water under over irrigation and over fertilization scenarios and efficient irrigation and fertilization scenarios were estimated. The results showed improvement of turf irrigation and fertilization management may decrease N-leaching significantly and greatly decrease the risk of ground water being contaminated by \( \text{NO}_3^-\)N leaching in the Salt Lake Valley.

(151 pages)
PUBLIC ABSTRACT

Characterizing Water and Nitrogen Dynamics in Urban/Suburban Landscapes

Hongyan Sun

This research investigated how to conserve water in urban landscapes and decrease nitrate leaching potential from urban landscapes to ground water. In the first study, we studied the water use of three types of plants—woody, herbaceous perennial, turf—from three putative water use classifications—Mesic, Mixed, Xeric. The plants were grown in large drainage lysimeters and each landscape plot was divided into woody plant, turf, and perennial hydrozones. Irrigation was applied for optimum landscape quality over two years, and water use was measured using a water balance approach. For woody plants and herbaceous perennials, we found that under well-watered conditions, canopy cover rather than plant type or water use classification was the key determinant of water use, which means homeowners and other landscape managers may achieve meaningful water savings by simply adjusting planting densities in the landscape. In the second study, a water and nitrogen transport model, Hydrus-1D, was calibrated and verified with observed soil water and soil nitrogen content data, and the calibrated model parameters were used to simulate nitrate leaching potential under over irrigation, over fertilization, and different soil texture scenarios. According to the simulation, under 1.5× irrigation and 2× fertilization levels, a 185.8 m² residential landscape would incur $570 in additional water expenses in an Intermountain Western city such as Denver, and potentially result in 743g N-leaching to ground water during a growing season. In the third study, based on simulated NO₃-N leaching from different soil textures obtained from the second study, a Geographic Information System (GIS) approach was used to estimate NO₃-N
leaching mass from urban turf areas. Risk maps of NO₃-N leaching were obtained for over irrigation and over fertilization scenarios and efficient irrigation and fertilization scenarios. The results showed improved irrigation and fertilization management may reduce turf NO₃-N leaching significantly, and result in more low NO₃-N leaching risk areas in urban areas of the Salt Lake Valley.
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Finally, I want to dedicate this dissertation to my husband, Jinqi Xue, and my family. Their love, care, patience, and big hearts led me to where I am today. They taught me tolerance, honesty, and wisdom. Family is what keeps me warm all the time.

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CHAPTER 1
INTRODUCTION

Drought and rapid population growth strain urban water supplies throughout the urbanizing Intermountain West (IMW) region of the US. Irrigated urban landscapes are the largest use of municipal water resources, and can consume approximately 60% of potable municipal water in the region. Because it is a limited resource in the IMW, efficient water use in irrigated urban landscapes is a fundamental long-term conservation policy for managing increasing demand and limited and uncertain supplies.

Water efficient landscaping is a key water conservation approach promoted in periodically water deficit regions of the United States. Water efficient landscaping can reduce water consumption without compromising landscape functionality or aesthetics. However, little research has quantified water needs of water efficient landscapes compared to traditional landscapes, particularly water use attributed to specific plant materials.

In addition to water, nitrogen (N) is a vital nutrient for enhancing plant growth, and soil conditions in the region often necessitate the application of water and fertilizers to meet landscape plant needs and to meet homeowner’s aesthetic expectations. However, homeowners often over-apply such amendments because of a lack of understanding of actual plant requirements. Excess water application in urban lawns can result in substantial NO$_3$-N leaching and overwatering in conjunction with fertilization can generate significantly higher NO$_3$-N loss in irrigated soils. NO$_3$-N is one of the common contaminants in ground water although it is necessary for human and environmental health since high NO$_3$-N concentrations in drinking water can be
harmful. Elevated NO$_3$-N concentration in drinking water can cause methemoglobinemia in infants and stomach cancer in adults. Ornamental landscaping takes up a large portion of land area in residential areas, and urban landscapes are composed of largely turfgrass, which may require extensive irrigation and fertilization. Therefore, fertilizer applied to urban lawns and gardens may be a potential source of NO$_3$-N to urban ground water, and may pose a negative impact on ground-water quality.

The leaching of NO$_3$-N from fertilizer applied to turfgrass depends highly on soil texture, N-source, fertilization rate and timing, the amount and timing of irrigation/rainfall, and the season of application. To determine the amount of NO$_3$-N leaching from plant root zones, NO$_3$-N concentration is often measured in leachate collected from suction lysimeters. However, this technique requires a great deal of replication to evaluate the multiple factors that may contribute to NO$_3$-N leaching. Therefore, models that predict flow and transport processes in soils are increasingly being applied to address practical problems. The use of simulation models allows extrapolation in time and space of data from leaching experiments and monitoring studies. In this study, a Hydrus-1D numerical model was utilized.

Since landscapes take up a large portion of residential areas and may receive a lot of fertilizer, there may be a high correlation between ground-water quality and residential areas around wells, as has been shown between agricultural land use activities and NO$_3$-N concentration in ground water. However, no correlation was determined between the percentage of residential land use surrounding the monitoring wells and the concentration of NO$_3$-N in water sampled from the wells in a USGS study in Salt Lake Valley. The absence of correlation between residential landscape area percentages and ground-water NO$_3$-N concentrations may be the result of
irrigation and fertilizers being applied only to turfgrass and ornamental plant areas rather than the entire residential area. Different soil textures under the landscapes may affect NO$_3$-N leaching as well. Therefore, a new approach integrating landscape areas, soil textures and different irrigation and fertilization scenarios may be more appropriate for estimating N-leaching from urban landscapes.

Furthermore, identification of areas with heavy NO$_3$-N leaching potential is important for land use planners and environmental regulators. Once such high-risk areas have been identified, preventive measures may be implemented to minimize the risk of NO$_3$-N leaching to ground water. Determination of the NO$_3$-N leaching risks of different areas may help to identify where ground water needs to be protected and where improved landscape management is needed. These determinations are also of importance in designating areas that can benefit from pollution prevention and monitoring programs.
CHAPTER 2

OBJECTIVES

The overall objective of this study was to determine water use of different plant types in landscapes based on water balances in drainage lysimeters, and to simulate water and N transport in turf, finally estimating potential NO$_3$-N leaching from urban landscapes.

To be specific, the objectives were to:

1. Develop water balances for water efficient landscapes with no soil water limits consisting of three putative water use characterizations—Mesic, Mixed and Xeric—and plant material of three different types—woody, herbaceous perennial, and turf—to develop K$_p$ values integrated at the irrigation zone and entire landscape level.

2. Utilize observed soil water content data, soil NH$_4$-N and NO$_3$-N data, and boundary condition assumptions for turfgrasses to calibrate the water and N transport process in the Hydrus-1D model, and to verify the model among different turf species.

3. Apply the calibrated and verified Hydrus-1D model to different turfgrass management scenarios (over irrigation, over fertilization, different soil textures) incorporating the potential factors that may affect N leaching.

4. Reanalyze the 1999 USGS ground-water NO$_3$-N concentration dataset and estimate NO$_3$-N turf leaching from different soil textures based on Hydrus-1D simulation to find the relationship between potential NO$_3$-N leaching from urban landscapes and ground-water NO$_3$-N concentration.

5. Find the high NO$_3$-N leaching risk areas in Salt Lake Valley that may pose potential risks to ground-water quality.
CHAPTER 3
WATER EFFICIENT URBAN LANDSCAPES – INTEGRATING DIFFERENT WATER USE CATEGORIZATION AND PLANT TYPES

Abstract. Little research has examined water requirements of entire irrigated urban landscapes integrating different types of plants. Three landscape treatments integrating different types of plants—woody, herbaceous perennial, turf—and putative water use classifications—Mesic, Mixed, Xeric—were grown in large drainage lysimeters. Each landscape plot was divided into woody plant, turf, and perennial hydrozones and irrigated for optimum water status over two years, and water use measured using a water balance approach. For woody plants and herbaceous perennials, canopy cover rather than plant type or water use classification was the key determinant of water use relative to reference evapotranspiration (ET$_o$) under well-watered conditions. For turf, monthly evapotranspiration (ET$_a$) followed a trend linearly related to ET$_o$. Monthly plant factors (K$_p$) for woody plants, perennials and turf species under well-watered conditions in this study ranged from 0.3 to 0.9, 0.2 to 0.5, 0.5 to 1.2, respectively. Adjusted K$_p$ for each hydrozone was calculated based on landscaped area covered by plant types as a percent of total area, and landscape factor (K$_l$) was calculated based on adjusted K$_p$ for each landscape treatment. Overall, K$_l$ relative to ET$_a$ ranged from 0.6 to 0.8 for three water use classifications.

1 Coauthored by: Hongyan Sun, Kelly Kopp, Roger Kjelgren
Drought and rapid population growth strain urban water supplies throughout the urbanizing Intermountain West (IMW). Irrigated urban landscapes are the largest use of municipal water resources, and can consume approximately 60% of potable municipal water in the region (Kjelgren et al., 2000; Utah Division of Water Resources, 2003). Because it is a limited resource in the IMW, efficient water use in irrigated urban landscapes is a fundamental long-term conservation policy for managing increasing demand and limited and uncertain supplies (St. Hilaire et al., 2008).

Xeriscaping, low water use landscaping, and water efficient landscaping, are key water conservation approaches promoted in periodically water deficit regions of the United States (Smith and St. Hilaire, 1999). In practice, these techniques are generally synonymous and refer to landscaping specifically designed to reduce water use relative to uniform turfgrass landscapes (St. Hilaire et al., 2008). For simplicity, this study will use the term water efficient landscaping to include mindful design, efficient irrigation systems, appropriate turf areas, appropriate plant material (turf and non-turf) choices, improved soil, mulching, and strategic maintenance.

Water efficient landscaping can reduce water consumption without compromising landscape functionality or aesthetics (St. Hilaire et al., 2008). However, little research has quantified water needs of water efficient landscapes compared to traditional landscapes, particularly regarding plant material. One 5-year study in Las Vegas, NV showed single-family homes with water efficient landscapes used 76% less water than turfgrass landscapes (Sovocool et al., 2006). However, those results were taken from a survey of voluntary participants such that traditional and water efficient landscapes differed in many ways including planting design, irrigation systems, and plant material. Since most water efficient landscaping principles apart
from plant material can be applied to any landscape, impact of plant selection alone is of research interest.

Plant water use characteristics inform designers, managers, and policy makers vested in water efficient landscapes. These stakeholders require information that allows estimation of minimum plant water demand that balances atmospheric evaporative pressure with visual, functional and health performance expectations (Shaw and Pittenger, 2004; White et al., 2004). The existing approach to estimating urban landscapes irrigation water use is derived from agriculture. The American Society of Civil Engineers Penman-Monteith (ASCE-PM) reference evapotranspiration ($ET_o$) equation is the simplified and accepted standard reference in agricultural settings for estimating plant water use with no soil water limits (Allen et al., 2005a). The ASCE-PM can model water used by a hypothetical reference short, cool-season grass surface based on inputs of local wind, air temperature, humidity, and incoming shortwave solar radiation. Calculated $ET_o$ is then corrected (for crops, typically but not always downward) with an empirical species-specific correction coefficient ($K_c$) that is a fraction of $ET_o$ such that:

$$ET_c = ET_o \times K_c$$  \hspace{1cm} (1)

where $ET_o$ is estimated plant water use proportional to irrigation water requirements for optimum, quantitative yield of a target crop. Equation 1 assumes vertical water movement controlled by stomatal opening and wind from a large, uniform crop surface, mirroring underlying assumptions, and thus a linear function, of $ET_o$ (Bos et al., 2008).

Many of these assumptions do not translate well to urban landscapes. Sufficient urban fetch and solar exposure for calculating $ET_o$ for a uniform plant surface complicate and limit weather station site selection (Eching and Snyder, 2005). Ideal
urban weather station sites with a uniform plant surface are then at odds with the non-uniformity, small size, and ventilated roughness characteristics of urban landscapes. Moreover, plants in urban landscapes are diverse architectural types—trees, shrubs, perennials, turfgrass—manifesting a wide range of water use characteristics. Further, urban landscape plants succeed when meeting appearance expectations rather than yielding a quantitative product. Biophysical diversity and appearance expectations suggest minimum needs in a water efficient landscape are a subjective threshold, rather than an objective target (Shaw and Pittenger, 2004). This threshold is potentially much lower than what plants would use with unlimited water supply, and may be achieved even when plants are water limited or stressed. Consequently, a plant factor $K_p$ (Eching and Snyder 2005; EPA WaterSense, 2009), rather than a coefficient $K_c$, more candidly represents the attenuated relationship between heterogenous urban landscape biophysical water use and homogenous urban $ET_o$. $K_p$ can characterize minimum water needs of general landscape plant types—woody and herbaceous—but can be species specific for the few commonly used turfgrass species, since turf $K_p$ may be equal to $K_c$ when grasses are well-watered and obtain optimum growth and development.

Species complexity in distinguishing minimum plant needs from maximum, well-watered use constrains development of landscape $K_p$ values useful to water efficient landscape stakeholders. Well-watered $K_p$ values for warm and cool season turfgrass species have been reasonably well characterized (Aronson et al., 1987; Carrow, 1995; Fry and Butler, 1989; Kopec et al., 1988), but minimum turfgrass water requirements have not. Plant factors have been reported for a number of landscape (tree, shrubs, herbaceous) species under well-watered (Beeson, 2005; Montague et al., 2004; Pannkuk et al., 2010) and minimum, water limited conditions (Pittenger and Henry,
These reports cover a small percentage of the total number of possible landscape plants, and how $K_p$ values developed in one climate translate to a different climate is problematic (Kjelgren et al., 2005).

Further complicating water efficient landscape water needs estimation, scaling an assemblage of $K_p$ values up to part of or the entire urban landscape is an increasingly necessary but conceptually muddled process. Increasing use of ET$_o$-based smart controllers demands input of a $K_p$ for turf and typically mixed species landscape plants for setting irrigation schedules at individual irrigation zone level. Policy needs for allocating a fixed amount of water to end users demands a $K_p$ over an entire landscape (often referenced as $K_l$; see Costello et al., 2000), for setting water allocation at policy level. Theoretical approaches to zone-level or landscape level have suggested assigning $K$ values grouped by plant types (tree, shrub, perennial, turf; EPA WaterSense, 2009; Water Use Efficiency Branch, 2009) or water use categorization (high-medium-low; Costello et al., 2000), each with various factors to correct for climate, plant density, and sometimes water stress (Bos et al., 2008; Eching and Snyder, 2005). However, there is little empirical data validating grouping of minimum water needs by plant type, water use categorization, or various correction factors (see Devitt and Morris, 2008; Pannkuk et al., 2010; Sachs et al., 1975).

Consequently, empirical data is needed to distinguish plant water use of different plant types and water use categorizations. This research was conducted under well-watered conditions in larger designed landscapes comprised of plant types such as turf, perennials, and woody plants. Once established, minimum water efficient landscape water needs under water-limiting conditions can then be more clearly defined. Objectives of this study were to develop water balances for water efficient landscapes.
with no soil water limits consisting of three putative water use characterizations—Mesic, Mixed, and Xeric—and plant material of three different types—woody, herbaceous perennial, and turf—to develop $K_p$ values integrated at irrigation zone and entire landscape level.

**Material and Methods**

1) Experimental site and design

This study was conducted at the Utah Botanical Center (UBC), Kaysville, UT, USA, latitude: 41°01’ N.21, longitude: 111°56’W. Annual precipitation averages 432 mm (Moller and Gillies, 2008), mostly as snow. The experimental site has a high mountain desert climate, with temperature extremes ranging from -30°C in January to 41°C in July. Average daily temperatures range from -4°C in January to 24°C in July. Soil is a Kidman fine sandy loam (coarse-loamy, Mixed, Mesic Calcic Haploxeroll) (USDA, 1968).

Experimental layout consisted of three different landscape treatments with three replicates (3 treatments × 3 replicates) installed in nine large drainage lysimeter plots (61.3 m$^3$, 9.14 m long × 6.1 m wide × 1.1 m deep each). For each lysimeter plot, surface was laser leveled to prevent horizontal surface water flow, while the bottom was graded at a 3% slope along its length, and then lined with a 4.5 mm thick pond liner. A 10.16 cm diam. perforated polyvinyl chloride (PVC) drain pipe encased in a silt sleeve was installed in a 2-3 cm diam. gravel bed at the low end to facilitate drainage to a collection well, as shown in Fig. 3-1, allowing monitoring of water quantity leaching through the soil profile of each lysimeter. Once lined and plumbed, subsoil and topsoil were returned to each plot and compacted to simulate original soil bulk densities of approximately 1.6 g cm$^{-3}$. 
Landscape treatments were assigned in a completely randomized block design (Fig. 3-1). Landscape treatments were three putative plant material water use classifications: Mesic (conventional landscape species), Xeric (native/adapted plant species of the IMW), and Mixed (both conventional and IMW native species), replicated three times. Each treatment lysimeter plot was divided into three hydrozones: woody plants, turf, and perennials (Fig. 3-1). Turf zones were bordered by steel edging 4 cm into soil and 2 cm above soil to prevent root growth outside turf area; this edging also prevented water from running off turf hydrozone. Planting plans for landscapes were spatially identical, differing only in plant species used in the putative water use classification treatments (Table 3-1). Plants were purchased from local retail nurseries and installed using accepted horticultural practices in 2004. Irrigation systems, soil properties, mulches, and maintenance practices were same for all plots. Bark mulch was applied to woody plant and perennial hydrozones to approximately 0.1 m depth to prevent soil water evaporation. Woody plants and perennials were pruned in June 2009 and May 2010 to facilitate plant growth and to ensure plants in the same treatment were similar in size, and pruned back to plot edge at beginning of each growing season. Fertilizer was applied to turf only due to the purpose of the entire project at a rate of 146.5 kg N/ha/year, divided into three applications in spring, midsummer and later fall; trees, shrubs, and perennials were not fertilized during study because fertilization to Xeric plants may lead to mortality and no nutrient stress symptom was observed for all the perennials and woody plants for the last 5 years before this study started. Mesic and Mixed turf were mowed weekly in 2009 and bi-weekly in 2010, while Xeric turf was mowed approximately once every 3 weeks in both years, mowing frequency changed due to labor availability.
Three 2.54 cm solenoid valves were installed (Rain Bird DV Series\(^2\), Rain Bird Corporation Azusa, CA) for each treatment lysimeter to distribute water according to woody, turf and perennial hydrozones. Pop-up sprinklers (15.24 cm) (Rain Bird 1800 Series) with 1.83 m variable arc nozzles (Rain Bird 6VAN) were installed in each turf hydrozone. Drip emitters (51 and 19.4 L/h) were installed for woody plant and perennial wildflower hydrozones in each treatment lysimeter, respectively. Emitters were distributed based on location of each plant (each plant was assigned 1 emitter), except for groundcovers and trees which had two or three emitters depending on their size.

2) Irrigation control

In each hydrozone, four Acclima soil moisture sensors (Acclima Inc. Meridian, ID, USA) were installed at 80, 45, 20, and 5 cm depth. Three Acclima CS3500 irrigation controllers were installed and each controller was connected to 36 sensors (3 plots × 3 hydrozones × 4 sensors) from one of Mesic, Mixed, and Xeric treatments representing a replicated block (Fig. 3-1). Three lysimeter plots in one of the blocks (one each of Mesic, Mixed, and Xeric landscapes) were chosen as master plots and connected to the master controller (Fig. 3-1) and the other two plots of each treatment were set as slaved plots under the same-treatment master plot. The Acclima CS3500 master controller was then used to control irrigation for all lysimeter plots. In master lysimeter plots, sensors at 5 cm for three turf hydrozones and 20 cm for three each of woody plant and perennial hydrozones were used to control irrigation in three plant types by detecting volumetric water content that exceeded set water level thresholds.

\(^2\) Mention of a trademark, proprietary product, or vendor does not constitute a guarantee or warranty of the product by the ASHS and does not imply its approval to the exclusion of other products or vendors that also may be suitable.
When volumetric soil moisture readings decreased below the set threshold, controlling sensors activated irrigation valves. Irrigation was stopped when soil water content readings of the controlling sensors reached field capacity (28.8%) for all three plant types. Soil moisture sensor readings in woody and perennial hydrozones were affected by proximity to nearby emitters. To avoid variation caused by emitter locations, soil moisture sensor readings were monitored weekly and emitter locations were adjusted weekly as needed.

Average field capacity is generally described by $\theta$ at -0.033 MPa matric potential ($\theta_{FC}$) and average permanent wilting point is generally described by $\theta$ at -1.5 MPa matric potential ($\theta_{PWP}$). The difference between $\theta_{FC}$ and $\theta_{PWP}$ is plant available water (PAW) (Blonquist et al., 2006). Threshold water content ($\theta_{\text{thresh}}$) is the water content level to which soil is allowed to dry before next irrigation event. Thus, $\theta_{\text{thresh}}$ lies between $\theta_{FC}$ and $\theta_{PWP}$ and can be established via selection of a management allowed depletion (MAD) value (Cuenca, 1989). The MAD is the percentage of PAW that can be extracted from plant root zone before irrigation is required, and can be used to calculate $\theta_{\text{thresh}}$:

$$\theta_{\text{thresh}} = \theta_{FC} - \text{MAD} \times (\theta_{FC} - \theta_{PWP})$$  \hspace{1cm} (2)

where all $\theta$ values are dimensionless values [L$^3$ L$^{-3}$] representing percentage of volume of water relative to total volume of soil considered. Cuenca (1989) reported MAD values of 33% for shallow rooted turf, 50% for medium rooted perennials, and 67% for deep-rooted woody plants.

In this study, MAD values of 33%, 50% and 67% were utilized for turf, perennial and woody plant hydrozones, respectively. Field capacity and $\theta_{PWP}$ were calculated with the van Genuchten (VG) water retention curve (van Genuchten, 1980). Threshold water content and field capacity for irrigation of each hydrozone were:
woody plants (14.2% and 28.8%); turf (22.4% and 28.8%); perennials (18.4% and 28.8%). To avoid overlapping irrigations and to make sure water pressure was the same for each hydrozone, the Acclima CS 3500 controller’s built in function “max zones watering simultaneously” was set to only one zone, and daily allowed irrigation time period was set between 8:00 PM to 8:00 AM to reduce daytime evaporation. While woody plant, turf, and perennial hydrozones were watered separately, each replicate was irrigated in the same manner and sequentially to achieve same volume of water application. For example, *Poa pratensis* L. in each Mesic landscape was watered sequentially, and received same total volume of water. Irrigation duration for each slave hydrozone could be adjusted as percent of master hydrozone to adjust difference in irrigation volume caused by sprinklers or emitters.

3) Data collection

a) Soil water content

In each hydrozone, four Acclima soil moisture sensors at depths of 80, 45, 20 and 5 cm measured volumetric soil water content data representing soil layers between 100-60 cm, 60-30 cm, 30-10 cm and 10-0 cm every hour. The three Acclima CS3500 irrigation controllers had datalogger capabilities and were utilized to log volumetric soil water content data hourly, and timing and duration of irrigation for each hydrozone of nine lysimeter plots were recorded by the master controller.

b) Leachate

Leachate from each landscape drained to collection wells adjacent to plots (Fig. 3-1), and was quantified using dipper trays connected to a CR1000 data logger (Campbell Scientific, Logan, UT, USA). Nine manual counters were connected to dripper trays as a back up to the data logger. Volume of each dip for each dipper tray
was calibrated before installation, and dipping times were recorded every 10 minutes and logged weekly, and manual counter data were collected weekly. Leachate volume was determined by the product of dipping volume and dipping times.

c) Irrigation data

A DLJ^3 1.905 cm flow meter (Daniel L. Jerman Co., Hackensack, NJ, USA) connected to a CR1000 data logger was installed in each plot and every 3.785 L of water applied to plots was recorded. Acclima CS3500 irrigation controllers recorded duration time of irrigation for each hydrozone, and the combination of irrigation volume and irrigation duration was used to determine amount of irrigation water applied to each hydrozone. If two zones in one lysimeter were irrigated one after another, duration was used to separate volume data.

d) Canopy cover estimation

In 2009, length and width of each plant in woody plant and perennial hydrozones were measured on mid-July and mid-September, and canopy cover of each plant type hydrozone was determined:

\[
\text{Canopy cover} = \sum \text{width} \times \text{length} / \text{area.} \tag{3}
\]

In 2010, a point-line intercept method (Salo et al., 2008) was used to estimate canopy cover monthly from May to October. With this method, canopy cover was measured along a linear transect line and was based on number of “hits” on a target plant out of the total number of points measured along that line. In each plot, spacing between lines was 0.61 m, and between points was 0.30 m.

e) Turf root distribution

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Measurements of effective turfgrass root length distribution were taken in May 2010. Soil samples from each turfgrass area were collected at depths of 0-10 cm, 10-20 cm, 20-30 cm, 30-40 cm, 40-60 cm, 60-80 cm using a soil auger, and soil was washed from roots in the lab. Root length density was measured by a modified line intersect method (Tennant, 1975). Roots were cut into 1 cm lengths and randomly placed into a transparent dish which was divided into 1×1cm squares. Instances of intersections of roots on both vertical and horizontal lines were counted.

Root length (R) = number of intercept (N) * length conversion factor. (4)

where conversion factor for 1 cm grid squares is 0.7857.

Root length density = root length / soil volume  (5)

After root length was measured, root samples were dried in an oven at 80 °C until a constant weight was reached, and dry weight of root samples were measured.

d) Weather data

In 2009, precipitation data was obtained from the UBC weather station, located 200 m from research plots. Reference ET data (UN-FAO 56) (Allen et al., 2005b) was obtained from the Farmington, UT weather station, located 2 miles from research plots due to a malfunction of the UBC weather station. In 2010, a TR-525i tipping bucket rain gauge (Campbell Scientific, Logan, UT, USA) was installed next to plots to collect precipitation data and ET₀ dataset was obtained from the UBC weather station.

4) Data analysis

a) ET calculation with water balance equation

Although it was not possible to separate drainage for each hydrozone in each lysimeter plot, drainage could be assigned to each hydrozone based on area and
reading of soil moisture sensors at 80 cm depth in each hydrozone. For example, in spring when drainage was greatest and caused by precipitation, it was assigned to each hydrozone based on area percentage of each hydrozone. In summer, over-irrigation rarely happened due to closely controlled irrigation within water thresholds by the smart irrigation controller, and precipitation was negligible, so there was almost no leaching from the plots. If leaching did occur during summer months, and there was no precipitation, leachate was assigned to the turf hydrozone as turf was irrigated most frequently and had the greatest deep-soil water content. If the situation was unclear, readings of soil moisture sensors at 80 cm in each hydrozone were the determining factor in assigning source of drainage. In addition to drainage data, irrigation data was available for each hydrozone, and water depth in soil profile of each hydrozone was obtained by daily averaging of hourly soil moisture data:

\[
\text{Soil profile water depth (mm) } = \theta_{80}\times 400 + \theta_{45}\times 300 + \theta_{20}\times 200 + \theta_{5}\times 100 \text{ (mm)}
\]  

(6)

As a result, monthly actual water use (ET\(_a\)) for each hydrozone during growing season was calculated by a water balance equation:

\[
\text{Monthly hydrozone ET}\_a = \Delta W + \text{Precipitation} + \text{Irrigation} - \text{Leachate} - \text{Runoff}
\]

(7)

where

\[
\Delta W (\text{water absorbed by plants}) = \text{initial water depth} - \text{final water depth}
\]

(8)

There was no runoff from the plots since each plot was lined to prevent water movement outside of the plot, thus runoff was set to zero.

b) \(K_p\)

\(K_p\) was calculated as:

\[
\text{Monthly } K_p = \text{Monthly ET}_a / \text{Monthly ET}_o
\]

(9)
Based on calculated monthly ET$_{a}$ (Eq. 7), seasonal ET for plant growing season from May to October was summed, and the sum of monthly ET$_{o}$ from May to October was denoted as seasonal ET$_{o}$:

\[ \text{Seasonal } K_p = \frac{\text{seasonal ET}_a}{\text{seasonal ET}_o} \]  
\[ (10) \]

c) Landscape coefficient ($K_l$)

Since landscapes are comprised of woody plants, turf and perennial hydrozones, and each group of plants has different $K_p$ values, and overall water use of each landscape depends on both $K_p$ and percent area of each hydrozone, area of each hydrozone was incorporated, thus obtaining adjusted $K_p$ for woody plants, turf, and perennials:

\[ \text{Adjusted } K_p = K_p \times \% \text{ area} \]

And $K_l$ of entire landscape is the sum of adjusted $K_p$ for each hydrozone:

\[ K_l = K_p^{\text{woody}} \times \% \text{ area} + K_p^{\text{turf}} \times \% \text{ area} + K_p^{\text{perennial}} \times \% \text{ area} \]
\[ (11) \]

d) Statistical analysis

To assess plant coverage on water use, plant factor ($K_p$) was regressed on percent canopy cover for each plant type and water use categorization over both years (Table Curve 2-D, Ver 5.01). This study was arranged in a completely randomized block design, and canopy cover, adjusted $K_p$ and $K_l$ data were analyzed using PROC GLM (Ver. 9.1, SAS Inc.\(^4\), Raleigh, NC, USA). When differences were significant, least-squares means (LSMEANS) tests were used to separate differences among means ($P=0.05$).

\[^4\] Mention of a trademark, proprietary product, or vendor does not constitute a guarantee or warranty of the product by the ASHS and does not imply its approval to the exclusion of other products or vendors that also may be suitable.
Results and Discussion

1) Weather conditions, soil water depletion, and irrigation timing for each hydrozone

In both years of the study, spring periods (April-June) were relatively cool and wet (Fig. 3-2). In 2009, greatest seasonal precipitation occurred in April, with cool wet conditions extending into June, resulting in significant soil water content peaks for all landscape treatments and irrigation zones in April (Fig. 3-3). In 2010, rainfall was generally continuous from April-June, greatest precipitation occurred in May, also resulting in soil water content peaks during that period (Fig. 3-2 and Fig. 3-3). Compared to historical ET and precipitation, in 2009, May had similar ET to historical ET with just 56% of historical precipitation, while June had 90% of historical ET and 199% of historical precipitation. In 2010, May was wetter than usual with only 73% of historical ET and 117% of historical precipitation, while June had 91% of historical ET and 88% of historical precipitation (Table 3-2). As air temperature and ET increased, and rainfall decreased in July-August both years (Fig. 3-2), plants depleted soil water storage. Woody plants resulted in rapid decreases while perennials resulted in slow decreases in soil water content in deep soil layers in each hydrozone in June and July of both years (Fig. 3-3).

In general, soil water content at 5 cm depth changed significantly following each irrigation application in each treatment while soil water content at 20 cm was less responsive to irrigation input compared to soil water content at 5 cm (Fig. 3-3). Irrigation water rarely reached deep soil layers (45 cm and 80 cm) because depth of application was regulated by shallow soil sensors for all plant types.

Overall, woody plant water consumption came primarily from water stored in soil in early summer, and irrigation only began to supply water when soil moisture
sensors at 20 cm detected lower water content threshold later in the growing season. In 2009, 62 mm of June rainfall forestalled irrigation onset for Mesic, Mixed, and Xeric woody treatments until July; in 2010, 27 mm of June rainfall initiated earlier soil water depletion and irrigation onset (Table 3-3, Fig. 3-3). In both years, Xeric woody plants depleted soil water at 20 cm more rapidly and initiated irrigation earlier than Mesic and Mixed treatments (Table 3-3). Once seasonal hot and dry conditions began both years, woody plants rapidly depleted deep soil water within the entire soil profile to a greater degree than turfgrasses or perennials. Unexpectedly, Xeric woody plants had greater irrigation frequency and greater deep soil water content than Mesic and Mixed woody plants in this study, suggesting more opportunistic root systems acclimated to an unlimited shallow water supply. By contrast, Mixed woody plants in 2010 did not deplete water to the point of triggering irrigation until early August (Table 3-3).

Perennials exhibited water depletion trends similar to woody plants, initiating irrigation earlier in 2010 than 2009. Perennials differed from woody plants in using less water deeper in soil profile (Table 3-3), as water contents at 80 cm were much greater than for woody plants (Fig 3-3). This suggests woody plants were deeper rooted with greater water uptake ability deeper in soil. In both 2009 and 2010, Mixed perennial plants had greater irrigation frequency than both Xeric and Mesic perennials, indicating shallow rooting for this particular configuration of plants compared to species used in the other two perennial water use classification treatments. Shallower root systems could account for greater irrigation of Mixed perennials compared to Mesic and Xeric perennials. Mesic perennials had similar irrigation depths to Mixed perennials in 2009, but much less irrigation frequency and irrigation depth in 2010. Unexpectedly low irrigation frequency and depth in Mesic perennials in 2010 may
have resulted from malfunction of the controlling soil moisture sensor in the master Mesic perennial hydrozone in 2010.

Turf irrigation was earlier than other plants types, but similar in starting later in 2009, at the beginning of May, compared to end of April 2010 (Table 3-3, Fig. 3-3). Since the irrigation-controlling soil moisture sensor was located at 5 cm and the threshold for turf was higher than that of woody plants and perennials, meaning less allowable water that could be depleted, this triggered greater watering frequency. Frequent irrigation promoted shallow rooting in upper 10 cm of soil (76%, 76% and 24% for Mesic, Mixed and Xeric, respectively, according to root length density; 88%, 88% and 53% for Mesic, Mixed and Xeric turf species, respectively, according to root dry weight). As a result, turf relied more on shallow soil water and had little encouragement to deplete deep soil water under well-watered conditions of this study. However, deep roots at 80 cm were observed but were rather sparse in all species (2%, 2%, and 15% for Mesic, Mixed, and Xeric, respectively, according to root length density; 1%, 1%, and 11% for Mesic, Mixed, and Xeric turf species, respectively, according to dry weight). Only Mesic-Kentucky bluegrass slowly depleted deep soil water over the growing season both years to less than 20%. For Mixed-tall fescue and Xeric-buffalograss, considered deep rooted and drought tolerant turf species (Carrow, 1996; Stewart et al., 2004), deep soil water content was at mid-20% throughout both growing seasons (Fig. 3-3), even when a Mesic turf irrigation valve malfunctioned in 2009 for a brief period.

This pattern of shallow rooting under frequent irrigation suggests Mixed-tall fescue and Xeric-buffalograss may have opportunistic root systems that preferentially use shallow water under frequent, shallow irrigation, relying on deep roots to exploit deep soil water when surface water supplies are depleted. Mesic-Kentucky bluegrass
appears to have a resourceful root strategy, depleting water throughout soil profile through dense shallow roots and sparse deep roots (Stewart et al., 2004), resulting in lower deep soil water content at the end of the growing season compared to the other turf species. However, under drought conditions, Mixed-tall fescue and Xeric-buffalograss turfgrasses may deplete deeper soil water after shallow soil moisture is depleted (Carrow, 1996), suggesting a small number of roots deep in soil may contribute substantially to a plant’s ability to avoid drought (Ervin and Koski, 1998).

2) Monthly ET and plant factor ($K_p$)-Woody plants and perennials

Monthly $E_{T_0}$ of each hydrozone was determined based on developed water balances (Fig. 3-4). Under well-watered conditions, $E_T$ of woody plants and perennials closely followed $E_{T_0}$ during growing season in both years of study (May to October). Previous research has found water use rates of many woody plant species may not closely follow $E_{T_0}$ because they are drought tolerant and can maintain acceptable aesthetic appearance under soil water deficits (Kjelgren et al., 2000), and in less humid climates, are susceptible to high-vapor-deficit induced stomatal closure and reduced transpiration (Kjelgren et al., 2005). Under well-watered conditions of this study, water use rates of integrated woody plants did closely follow $E_{T_0}$.

Actual $E_T$ of woody plants was lowest in May, increased in June, reached a peak in July or August, depending on treatment, and decreased in September and October (Fig. 3-4). Generally, Mixed woody plants had lesser $E_T$ than Mesic and Xeric woody plants, while Mesic woody plants had greatest $E_T$ among the three treatments. This finding is likely the result of different woody plant canopy covers (Table 3-3). Mesic woody plants had greatest canopy cover in both 2009 and 2010, while Mixed woody plants had least canopy cover in both years (Table 3-3).
Mesic, Mixed and Xeric perennials had much lesser ET$_a$ than ET$_o$ and ET$_o$ of the other two plant types during study (Fig. 3-4). Actual ET of Mesic, Mixed, and Xeric perennials, however, were very similar in May, June, September, and October of 2009. Additionally, there were no differences in canopy cover observed in 2009 among Mesic, Mixed, and Xeric perennials (Fig. 3-4, Table 3-3). In 2010, ET$_a$ of the three perennial treatments generally followed the trend of ET$_o$ (Fig. 3-4). Xeric perennials had the lowest ET$_a$ in June and lesser ET$_a$ than Mixed perennials in July. Xeric perennials also had less canopy cover than Mesic and Mixed perennials (Fig. 3-4, Table 3-3). Mesic perennials had greater canopy cover than Xeric perennials in 2010 (Table 3-3). However, ET$_a$ of the two treatments was similar in July 2010 (Fig. 3-4). This finding is likely due to an unintended water deficit that occurred in July 2010 when a controlling moisture sensor malfunctioned in the Mesic perennial hydrozone.

Plant factors combined over a range of woody plants in this study ranged from 0.2 to 1.0, and varied from month to month (Fig. 3-5). Generally, woody plants had lesser $K_p$ values at the beginning of the growing season and reached greater $K_p$ values during late growing season. This finding was likely the result of increasing canopy cover, and suggests a close relationship between canopy cover and water use in the landscapes studied. For perennials, $K_p$ values were lesser than woody plants (0.4 vs. 0.7 on average), and likely resulted from less canopy cover in perennial hydrozones (Fig. 3-5, Table 3-3). $K_p$ values for non-turf landscape plants have not been widely examined because of great species diversity and difficulty in quantifying $K_p$ values. For many woody species, stomatal sensitivity to high vapor pressure deficits and close coupling to atmospheric conditions result in a declining rate of water loss at high ET$_o$ rates (Buwalda and Lenz, 1995). Such nonlinearity suggests a wide range of $K_p$ for woody plants, depending on ET$_o$ conditions. For example, coefficients ranging from
0.2 to 0.8 of $E_T$ have been suggested for woody plants (Buwalda and Lenz, 1995), and from 0.2 to 1 have been observed in a range of broadleaf tree species (Montague et al., 2004).

The importance of canopy cover is illustrated in Fig. 3-6. A linear relationship between $K_p$ and percent canopy cover ($r^2=0.88$) of woody plants and perennials was found when both years of study were combined, indicating canopy cover was the controlling factor for water use of non-turf plants under well-watered conditions. Turf $K_p$ values are included as reference points (Fig. 3-6).

3) Monthly ET and plant factor ($K_p$)-Turf

Actual monthly evapotranspiration ($E_{Ta}$) of all turf species was close to $E_T$ and followed the same trend as $E_T$ (Fig. 3-4) although month-to-month variation was high, as has been reported for turfgrass (Carrow, 1995). Mixed-tall fescue had greater $E_{Ta}$ than $E_T$ in October of 2009 and during the first four months of the growing season in 2010. We suspect mis-alignment between the soil water sensor and irrigation may have triggered greater irrigation frequency than was warranted by actual turf water use. Mesic-Kentucky bluegrass $E_{Ta}$ was nearly same as $E_T$ during every month of the two years, except in August of 2010. Xeric-buffalograss had lesser $E_{Ta}$ than $E_T$ in early season of both years, possibly due to cooler conditions delaying full development of this $C_4$ species.

A variety of factors affect turf $K_p$ values: turf type ($C_3$ cool vs. $C_4$ warm season grasses), turf quality, stage of development, and to a lesser degree, turf height (Brown and Kopec, 2000). As a general rule, $C_3$ water use is greater than $C_4$, and in this study Xeric-warm season species used 8% and 11% less irrigation than Mesic and Mixed cool season species in 2009, and 21% and 38% less irrigation than Mesic and Mixed
cool season species in 2010. Overall $K_p$ values for Mesic and Mixed-cool season species were greater than Xeric-warm season species in early 2009 and all of 2010 (Fig. 3-5). However, in some cases warm-season grass water use may approach cool-season grass water use rates under well-watered conditions (Brown et al., 2001; Devitt et al., 1992; Jia et al., 2009). In this study, similar $ET_a$ and $K_p$ of Mesic, Mixed, and Xeric grasses were observed in July and August of 2009 (Fig. 3-4 and Fig. 3-5). Of the three plant types, turf $K_p$ has been studied most, and $K_p$ values observed in this study were comparable to previous research. For example, values of $K_p$ reported for cool-season turfgrasses range from 0.72 to 1.23 of $ET_o$ (Aronson et al., 1987), compared to 0.6 to 1.2 in our research, while those for warm-season turfgrasses range from 0.67 to 0.84, compared to from 0.5 to 1.0 in our research (Carrow, 1995).

4) Plant factors integrated by zone and landscape

Adjusted $K_p$ for each hydrozone were calculated based on landscaped area covered by plant types studied as a percent of total area (Table 3-4). The percent of total landscape area covered by woody plants, turfgrass, and perennials in this study totaled 43%, 35%, and 22%, respectively. Mesic and Xeric woody plants had similar adjusted $K_p$ values that were greater than adjusted $K_p$ for Mixed woody plants (because of greater $K_p$). However, perennial adjusted $K_p$ were very similar for all three landscape treatments in both 2009 and 2010 because their water use was similar, and lesser than other plant types due to less canopy cover and lower percent area. For turfgrasses, Xeric-buffalograss had lesser adjusted $K_p$ than Mesic-Kentucky bluegrass and Mixed-tall fescue species, again because of lesser water use. Turf adjusted $K_p$ values were greater than that of woody plants and perennials as a result of high turf $K_p$ and relatively large turf canopy areas in landscapes. Although adjusted $K_p$ values for
each hydrozone cannot be used as a guideline for irrigation control in landscapes, they do reflect differences in canopy cover and plant types useful for assessing water conservation of an entire landscape. In the case of this study, plant types refers to turf versus non-turf plants with potentially variable canopy cover, rather than woody, perennial or ground cover types previously suggested (Bos et al., 2008; Eching and Snyder, 2005; EPA WaterSense, 2009).

Overall, $K_l$ of the landscapes ranged from 0.6 to 0.8 under well-watered conditions of this study (Table 3-4). Mesic landscape had the highest $K_l$ for both years of study. Mixed landscape had lesser $K_l$ in 2009, while Xeric landscape had lesser $K_l$ in 2010. Landscape factors can be used as a tool in irrigation decision-making, which could contribute to water savings in amenity landscapes (Pannkuk et al., 2010). Few $K_l$ values have been reported, although Pannkuk et al. (2010) reported $K_l$ of St. Augustine grass ranged from 0.45 to 0.62 seasonally, and for mixed-species landscapes ranged from 0.5 to 0.7 in southern Texas. Landscape factors for complete landscapes, including woody plants, turf and perennials have not been previously reported. Therefore, $K_l$ values developed under well-watered conditions in this study may provide guidelines as exploratory standards in allocating landscape irrigation water in the IMW region.

This study suggests classifying plants by type (height) or water use may not be useful for estimating water demand and irrigation management of water efficient landscaping. Absence of $K_p$ differences among plant water use categorizations indicates the perception of drought tolerant plants being low water use plants needs to be clarified. For example, woody and perennial species in this study categorized as low water use from arid-xeric habitats consumed almost as much water as Mesic plants under well-watered conditions, differing only by canopy cover fraction (Fig. 3-
6). High water use rates in plants from dry habitats is not surprising, as many have deep roots to forestall water stress, but many also have shallow roots for scavenging surface water from unpredictable summer rain. *Pinus edulis*, used in this study and widespread in the IMW, has been shown to respond to shallow surface watering in addition to deep roots to tolerate drought characteristic of the region (West et al., 2007). The same mechanism appears to apply to tall fescue used in this study. So the three putative water use classifications—Mesic, Mixed, Xeric—are perhaps better described as differences in drought tolerance, or minimum water needs classifications.

Ability to tolerate low soil water conditions varies widely among species and may be considered as a drought tolerance rating, meaning minimum level of plant water needed to achieve an acceptable appearance in a landscape. Therefore, managing water efficient landscapes under certain levels of water stress may be possible while maintaining an acceptable appearance as well as achieving the objective of water conservation in landscapes since appropriate species are able to tolerate low soil water conditions (Montague et al., 2004; Reid and Oki, 2008). An advantage of water efficient landscaping irrigated at minimum water needs is a reduction of luxury water use. Since ornamental landscapes are valued for their appearance rather than growth or yield, maximum well-watered irrigation and resultant luxury plant water use may result in greater vegetative growth and, consequently, more pruning and mowing, increasing labor as well as water costs. Another benefit of irrigating to minimum water needs for more xeric plants is encouraging deeper rooting and exploiting a greater volume of soil water during dry periods.

The trend (Fig. 3-6) indicating similar turf and non-turf $K_p$ at well-watered, full canopy cover suggests assigning different $K_p$ values to non-turf landscape plants based on type varying by height (tree, shrub, ground cover) is probably not a
meaningful distinction in water efficient landscaping. Similar well-watered, full canopy water use rates between woody and turf plants are likely a trade-off between boundary layer and stomatal limitations. Woody plant zones in this study presented a rough (variable height), well ventilated canopy closely coupled to atmosphere (Seraphin and Guyenne, 2008), even the near complete canopy cover Mesic zone, compared to turf. Consequently, high vapor deficits characteristic of arid regions (Gao et al., 2005) are imposed at leaf level (Jarvis, 1985) typically trigger stomatal closure (Turner et al., 1984) that increases with even small changes in plant height (Medeiros and Pockman, 2010). Stomatal sensitivity to vapor deficits is common in woody plants, moderating transpiration rates (Choudhury and Monteith, 1986) when leaf area indices (LAI) are similar to turf (Pereira et al., 2007). While woody plant canopies can reach high LAI (Schleppi et al., 2011) up to twice that of turf (Pereira et al., 2007), ventilation and stomatal sensitivity to vapor deficits also increases with height (Ambrose et al., 2010), again moderating transpiration rates (Choudhury and Monteith, 1986). The tradeoff is turfgrass may have high stomatal conductance rates but overall canopy transpiration is limited by low boundary layer conductance (Jarvis, 1985).

A potentially more meaningful distinction would be adjusting \( K_p \) values based on canopy cover (Fig. 3-6). Intuitively, water loss decreases if number of transpiring leaves decreases within a given area such as an irrigation zone. Figure 3-6 indicates \( K_p \) can be adjusted downward almost at a 1:1 basis as percent cover decreases, and sprinkler irrigation application frequency can be adjusted accordingly. Below 50% canopy cover drip/low volume irrigation is a more water efficient choice where number of leaves would also be the primary driver of water needed by an individual plant relative to \( K_p \).
Modification of $K_p$ by canopy cover and drought tolerance rating leads to adjusting downward well-watered $K_p$ in landscape irrigation zones of mixed plant types. This would enable landscape managers and designers to achieve greater water conservation when there is reduced canopy cover, low plant densities, and plants have known drought tolerance abilities. This approach appears feasible based on our findings and those reported under water limiting conditions (Pittenger and Henry, 2005; Reid and Oki, 2008; Shaw and Pittenger, 2004). Plants with greater drought tolerance, or lower water needs rating, may be of importance for use in water efficient landscaping. A turf $K_p$ with a common 100% plant cover would be controlled by turf drought tolerance abilities. For non-turf plants, however, canopy cover appears to be the controlling factor of water use under well-watered conditions. The $K_p$ is a function of canopy cover fraction, so the value of $K_p$ could be reduced by some function of canopy cover and species minimum water needs rating. The percent to reduce $K_p$ for a non-turf plant zone could be roughly estimated visually by a landscape manager, based on canopy cover and plant drought tolerance classifications, but minimum water needs would have to be carefully evaluated at the design stage, and mixing species of different minimum water needs would limit water conservation potential. A percentage reduction in zone $K_p$ value, programmed into smart irrigation controllers or station runtime, could be adjusted by irrigation manager using the global percentage function present in most irrigation controllers.

**Conclusions**

Under well-watered conditions of this study, we determined plant canopy cover—rather than plant material water use categorization—was the controlling factor in woody plant and perennial water use. This suggests that categorizing water use based
on plant type, as suggested by the EPA (EPA Water Sense, 2009) appears to not have merit. Consequently, landscape managers may achieve meaningful water savings by simply adjusting landscape-planting densities. In the meantime, adjusting percentage of landscape area devoted to woody plants, turf and perennials based on $K_p$ and adjusted $K_p$ of each hydrozone may provide another method for conserving water in landscapes under well-watered conditions. The $K_p$ values and irrigation timings for different plant types developed from this study may also serve as a guideline for setting well-watered irrigation schedules in the IMW region. Under water-stressed conditions, however, plant material choice will likely play a more central role in overall landscape water use. Plants with greater drought tolerance, or lower water needs rating, may be of importance for use in water efficient landscaping. The results of this study also suggest that mild water stress promotes water uptake deeper in the root zone, particularly for drought-adapted plants that have opportunistic water uptake patterns. Further research on water-stressed conditions is needed to ascertain drought tolerance for different plant types under different minimum water demand categorizations.

**Literature Cited**


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development of arid lands. New Mexico J. Sci. 38. New Mexico Acad. Sci., Albuquerque, NM.


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<http://www.water.ca.gov/wateruseefficiency/sb7/>.


Table 3-1. Plant list for Mesic, Mixed, and Xeric landscapes.

<table>
<thead>
<tr>
<th>Plant Type</th>
<th>Mesic</th>
<th>Mixed</th>
<th>Xeric</th>
</tr>
</thead>
<tbody>
<tr>
<td>Woody</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Evergreen tree</td>
<td><em>Pinus heldrichi</em> ‘Leucodermis’</td>
<td><em>Pinus aristata</em></td>
<td><em>Pinus edulus</em></td>
</tr>
<tr>
<td>Broadleaf evergreen</td>
<td><em>Buxus microphylla koreana</em></td>
<td><em>Euonymus kiautschovicus</em></td>
<td><em>Arctostaphylos coloradoensis</em></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>‘Panchito’</td>
</tr>
<tr>
<td>Evergreen shrub</td>
<td><em>Thuja occidentalis</em> ‘Little Giant’</td>
<td><em>Pinus mugo</em> ‘Pumilio’</td>
<td><em>Mahonia repens</em></td>
</tr>
<tr>
<td>Deciduous shrubs</td>
<td><em>Spiraea bumalda</em> ‘Anthony Water’</td>
<td><em>Syringa meyeri</em></td>
<td><em>Potentilla fruticosa</em></td>
</tr>
<tr>
<td></td>
<td><em>Euonymus alatus</em> ‘Compactus’</td>
<td><em>Berberis thunbergii</em></td>
<td><em>Purshia tridentata</em></td>
</tr>
<tr>
<td></td>
<td><em>Cornus sericea</em> ‘Kelseii’</td>
<td><em>Viburnum juddii</em></td>
<td><em>Chamaebiatia millefolium</em></td>
</tr>
<tr>
<td></td>
<td><em>Salix purpurea</em> ‘Nana’</td>
<td><em>Caragana arborescens</em></td>
<td><em>Fallugia paradoxa</em></td>
</tr>
<tr>
<td>Perennial</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Herbaceous perennial</td>
<td><em>Paeonia lactiflora</em> ‘Nippon Beauty’</td>
<td><em>Penstemon digitalis</em></td>
<td><em>Penstemon strictus</em></td>
</tr>
<tr>
<td></td>
<td><em>Hemerocallis hybrids</em></td>
<td><em>Sedum spectabile</em></td>
<td><em>Sphaeralcea grossulariaefolia</em></td>
</tr>
<tr>
<td></td>
<td><em>Salvia x superb</em></td>
<td><em>Lavandula angustifolia</em></td>
<td><em>Artemisia ludoviciana</em> ‘Silver King’</td>
</tr>
<tr>
<td></td>
<td><em>Phlox subulata</em> ‘Emerald Cushion Blue’</td>
<td><em>Oenothera missouriensis</em></td>
<td><em>Eriogonum corymbosum</em></td>
</tr>
<tr>
<td></td>
<td><em>Chrysanthemum superbum</em></td>
<td><em>Rudbeckia occidentalis</em></td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Aster novae-angliae</em> ‘Purple Dome’</td>
<td><em>Gaara lindehneri</em> ‘Siskiyou Pink’</td>
<td></td>
</tr>
<tr>
<td>Ground cover</td>
<td><em>Thymus pseudolanuginosus</em></td>
<td><em>Sedum spurium</em></td>
<td><em>Penstemon pinifolius</em></td>
</tr>
<tr>
<td></td>
<td><em>Vinca minor</em></td>
<td><em>Delosperma floribundum</em></td>
<td><em>Antennaria microphylla</em></td>
</tr>
<tr>
<td>Turf</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Long grass</td>
<td><em>Miscanthus sinensis</em></td>
<td><em>Calamagrostis acutiflora</em></td>
<td><em>Elymus cinereus</em></td>
</tr>
<tr>
<td>Short grass</td>
<td><em>Helictotrichon sempervirens</em></td>
<td><em>Festuca ovina glauca</em></td>
<td><em>Festuca idahoensis</em></td>
</tr>
<tr>
<td></td>
<td><em>Poa pratensis</em> L.</td>
<td><em>Festuca arundinacea</em> Schreb.</td>
<td><em>Buchloë dactyloides</em></td>
</tr>
</tbody>
</table>
Table 3-2. Comparison of monthly ETo and precipitation to monthly historical ETo and precipitation in 2009 and 2010 (Kaysville, UT).

<table>
<thead>
<tr>
<th></th>
<th>Historical</th>
<th>2009</th>
<th>2010</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>ETo (mm)</td>
<td>P (mm)</td>
<td>ETo/his. ET</td>
</tr>
<tr>
<td>May</td>
<td>148</td>
<td>74</td>
<td>149</td>
</tr>
<tr>
<td>Jun</td>
<td>178</td>
<td>31</td>
<td>159</td>
</tr>
<tr>
<td>Jul</td>
<td>207</td>
<td>23</td>
<td>198</td>
</tr>
<tr>
<td>Aug</td>
<td>177</td>
<td>23</td>
<td>175</td>
</tr>
<tr>
<td>Sep</td>
<td>118</td>
<td>35</td>
<td>122</td>
</tr>
<tr>
<td>Oct</td>
<td>71</td>
<td>54</td>
<td>56</td>
</tr>
</tbody>
</table>
Table 3-3. Irrigation frequency, start date, duration days, depth, soil water use, total water use, ratio of irrigation and soil water use, canopy cover, and seasonal $K_p$ for woody, turf and perennial hydrozones in Mesic, Mixed, and Xeric landscapes in 2009 and 2010 (Kaysville, UT).

<table>
<thead>
<tr>
<th></th>
<th>2009</th>
<th>2010</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Irrigation</td>
<td>Total</td>
</tr>
<tr>
<td></td>
<td>Events /year</td>
<td>Soil Water Use</td>
</tr>
<tr>
<td></td>
<td>Start Date</td>
<td>Duration Days</td>
</tr>
<tr>
<td>Woody</td>
<td>Mesic</td>
<td>22 22-Jul 74</td>
</tr>
<tr>
<td></td>
<td>Mixed</td>
<td>11 26-Jul 63</td>
</tr>
<tr>
<td></td>
<td>Xeric</td>
<td>26 3-Jul 87</td>
</tr>
<tr>
<td>Perennial</td>
<td>Mesic</td>
<td>12 20-Jul 65</td>
</tr>
<tr>
<td></td>
<td>Mixed</td>
<td>19 28-Jun 91</td>
</tr>
<tr>
<td></td>
<td>Xeric</td>
<td>7 28-Jul 52</td>
</tr>
<tr>
<td>Turf</td>
<td>Mesic</td>
<td>37 3-May 147</td>
</tr>
<tr>
<td></td>
<td>Mixed</td>
<td>44 3-May 161</td>
</tr>
<tr>
<td></td>
<td>Xeric</td>
<td>33 3-May 149</td>
</tr>
<tr>
<td></td>
<td>30-Jun 110</td>
<td>440 a 139 a 748 a</td>
</tr>
<tr>
<td></td>
<td>10-Aug 56</td>
<td>146 b 142 a 446 b</td>
</tr>
</tbody>
</table>
Leachate was subtracted and rainfall was added to the total water use. 184 mm and 215 mm rainfall occurred in the growing season of 2009 and 2010, respectively.

Data within a column of each year (2009 and 2010) and each plant type (woody, perennial, and turf) not followed by the same letter are different at $P \leq 0.05$.

<table>
<thead>
<tr>
<th>Type</th>
<th>Year</th>
<th>Marker</th>
<th>Date</th>
<th>Value 1</th>
<th>Value 2</th>
<th>Value 3</th>
<th>Value 4</th>
<th>Value 5</th>
<th>Value 6</th>
</tr>
</thead>
<tbody>
<tr>
<td>Xeric</td>
<td>34</td>
<td>7-Jun</td>
<td>134</td>
<td>455 a</td>
<td>75 b</td>
<td>690 a</td>
<td>6.1</td>
<td>0.76 b</td>
<td>0.81</td>
</tr>
<tr>
<td>Perennial</td>
<td>13</td>
<td>7-Jul</td>
<td>95</td>
<td>89 c</td>
<td>98 a</td>
<td>353 b</td>
<td>0.9</td>
<td>0.64 a</td>
<td>0.41</td>
</tr>
<tr>
<td>Mixed</td>
<td>35</td>
<td>16-Jun</td>
<td>122</td>
<td>248 a</td>
<td>48 a</td>
<td>446 a</td>
<td>5.2</td>
<td>0.53ab</td>
<td>0.52</td>
</tr>
<tr>
<td>Xeric</td>
<td>11</td>
<td>5-Jul</td>
<td>83</td>
<td>135 b</td>
<td>69 a</td>
<td>363 b</td>
<td>2.0</td>
<td>0.43bc</td>
<td>0.42</td>
</tr>
<tr>
<td>Turf</td>
<td>Mesic</td>
<td>44</td>
<td>27-Apr</td>
<td>170</td>
<td>667 b</td>
<td>53 a</td>
<td>870 b</td>
<td>12.6</td>
<td>1 a</td>
</tr>
<tr>
<td>Mixed</td>
<td>61</td>
<td>27-Apr</td>
<td>171</td>
<td>854 a</td>
<td>32 a</td>
<td>983 a</td>
<td>26.7</td>
<td>1 a</td>
<td>1.15</td>
</tr>
<tr>
<td>Xeric</td>
<td>39</td>
<td>9-May</td>
<td>165</td>
<td>530 c</td>
<td>30 a</td>
<td>658 c</td>
<td>17.7</td>
<td>1 a</td>
<td>0.77</td>
</tr>
</tbody>
</table>


Table 3-4. Adjusted $K_P$ based on percent of area for woody plant, turf, and perennial hydrozones and total $K_L$ of landscapes in Mesic, Mixed, and Xeric landscapes in 2009 and 2010 (Kaysville, UT).

<table>
<thead>
<tr>
<th>% Area</th>
<th>Adjusted $K_P$</th>
<th>Total $K_L$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Woody 43%</td>
<td>Turf 35%</td>
</tr>
<tr>
<td>2009</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mesic</td>
<td>0.30 bc</td>
<td>0.30 c</td>
</tr>
<tr>
<td>Mixed</td>
<td>0.22 d</td>
<td>0.31 c</td>
</tr>
<tr>
<td>Xeric</td>
<td>0.29 c</td>
<td>0.29 cd</td>
</tr>
<tr>
<td>2010</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mesic</td>
<td>0.38 a</td>
<td>0.36 b</td>
</tr>
<tr>
<td>Mixed</td>
<td>0.22 d</td>
<td>0.40 a</td>
</tr>
<tr>
<td>Xeric</td>
<td>0.35 ab</td>
<td>0.27 d</td>
</tr>
</tbody>
</table>

Means within a column not followed by the same letter are different at $P \leq 0.05$. 
Fig. 3-1. Diagram of the plots including the size and conceptual design of each plot, landscape treatments, and the location of sensors and controllers, collection wells, drainage pipes and drainage trench.
Fig. 3-2. Reference evapotranspiration (ET0), precipitation and average temperature from April to October in 2009 and 2010 (Kaysville, UT).
Fig. 3-3. Water depletion of soils (n=3) at 5, 20, 45 and 80 cm under woody plant, herbaceous perennial and turf plant types in Mesic, Mixed, and Xeric landscapes from April to October in 2009 and 2010 (Kaysville, UT).
Fig. 3-4. Monthly evapotranspiration (ETa) (mean ± SE, n=3) of woody plants, turf and perennials in Mesic, Mixed, and Xeric landscapes constructed in large drainage lysimeters from May to October in 2009 and 2010 (Kaysville, UT).
Fig. 3-5. $K_P$ (mean ± SE, n=3) of woody plants, turf and perennials for Mesic, Mixed, and Xeric landscapes constructed in large lysimeters over the growing season from May to September in 2009 and 2010 (Kaysville, UT).
Fig. 3-6. Relationship between seasonal $K_p$ (plant water use as a fraction of ETo) and canopy cover for woody plants, and perennials in Mesic, Mixed, and Xeric landscapes in 2009 and 2010 (Kaysville, UT). Large points show data in 2009 and small point show data in 2010. Turf $K_p$ values are included as reference points. Mesic and Mixed turf $K_p$ values overlapped in 2009.
CHAPTER 4
NUMERICAL SIMULATION OF WATER AND NITROGEN
TRANSPORT IN THREE TURF SPECIES

ABSTRACT

Nitrogen (N) leaching and contamination of surface and ground water is an environmental issue all over the world because of over irrigation and fertilization of agricultural lands, as well as turfgrass areas. In this study, water transport parameters were calibrated using an inverse simulation with Kentucky bluegrass (KBG). Subsequently those parameters were applied to simulate water use by tall fescue (TF) and buffalograss (BG) turfgrasses using numerical modeling (Hydrus-1D). Using the calibrated soil hydraulic parameters obtained from the water transport simulation, N transport and transformation was modeled with Hydrus-1D. A variable boundary condition was used to describe irrigation and fertilization schedules, including the composition of applied fertilizer and the estimated N content of returned turf clippings. Numerical simulations using multiple fertilizer application events at two different rates (1×, 2×) of Utah State University Cooperative Extension fertilizer recommendations and three irrigation levels (1×, 1.5×, and 2×) of optimized 2010 irrigation application for each turfgrass species were used to compare N leaching from the root zone. To demonstrate the influence of soil texture on N leaching, three different soil textures were also used in simulations. According to the simulations, under 1.5× irrigation and 2× fertilization levels, a 185.8 m² residential landscape would result in $570 additional water expense in Denver, and potentially result in 743 g N-leaching to ground water during a growing season. Under recommended

1 Coauthored by: Hongyan Sun, Kelly Kopp, Scott Jones

Comment [KK1]: Spaces in between the number and the ”×” or not? Be consistent throughout.
irrigation and fertilization schedules, 12% of applied N could be leaching to ground water while 41% and 62% of applied N could be leached to ground water under 1.5× irrigation + 2× fertilization, and 2× irrigation + 2× fertilization scenarios, respectively.

Nitrogen is a common surface water and ground water contaminant that can cause health problems in infants and animals, as well as the eutrophication of water bodies (Fennessy and Cronk, 1997). Global application of N fertilizer is equally distributed between developed and developing countries (Riley et al., 2001). As a result, N leaching from agricultural lands, as well as managed landscapes, which are receiving N fertilizers, may be an important issue all over the world. Furthermore, Galloway et al. (1995) estimated that global N fertilizer production will increase 60-90% by the year 2025. If the efficiency of fertilizer use is not increased, these N fertilizer applications will result in increased N losses as leachate to freshwater and marine systems (Riley et al., 2001). Managed landscapes planted with turfgrass may require regular irrigation and fertilizer applications, and are often perceived to be a source of N leaching, especially on coarse-textured soils (Sharma et al., 1996; Wakida and Lerner, 2005).

There is also public concern that fertilization of turfgrass systems, particularly additions of N on golf courses, may be adversely affecting ground-water quality due to nitrate (NO$_3$-N) leaching (Lee et al., 2003). Research reviewed by Petrovic (1990) suggested that NO$_3$-N in turf areas has the potential to leach through soils and contaminate ground water if not properly applied. In addition, the results of a lysimeter study showed that N application rates of 50 kg ha$^{-1}$ N on golf greens could create adverse environmental impacts on surface waters and ground water due to leaching losses of NO$_3$-N (Wong et al., 1998). And the use of fertilizers on
recreational landscapes such as golf courses has been identified as one of the sources of NO$_3$-N in urban aquifers (Sharma et al., 1996; Wong et al., 1998), as well as lawn fertilization in residential areas (Kopp and Guillard, 2005; Saha et al., 2007).

However, other research suggests that N leaching is not a problem in turf. Lee et al. (2003) found no evidence that N fertilization or the ecology of a bermudagrass system posed inherent risks to water quality and the environment. Cisar et al. (2004) suggested that turfgrass is relatively efficient at using applied N and, when properly maintained, offers minimal environmental impact compared to mixed species landscapes. The research of Wu et al. (2007) indicated that if turfgrass was properly managed, it provided an opportunity to mitigate NO$_3$-N loading to surface and ground water, even when N application rates were as high as 488 kg ha$^{-1}$ year$^{-1}$. Risks of NO$_3$-N losses in bermudagrass were avoidable with proper fertilization and irrigation programs, even when a highly soluble N source was used (Quiroga-Garza et al., 2001). Some research has also shown that properly applied fertilizer is mostly assimilated by the grass (Erickson et al., 2001; Snyder et al., 1984). Miltner et al. (1996) used N$^{15}$ labeled urea in Kentucky bluegrass (Poa pratensis L.) and found that a well-maintained turfgrass could intercept and immobilize N quickly, making leaching unlikely.

Although previous research has come to opposite conclusions regarding N leaching from turf, it is clear that fertilization may result in problematic N-leaching when the turf is not well managed. Excess water application in urban lawns can result in substantial NO$_3$-N leaching (Exner et al., 1991) and overwatering in conjunction with fertilization can generate significantly higher NO$_3$-N loss in irrigated soils (Morton et al., 1988). Overall, the leaching of NO$_3$-N from fertilizer applied to turfgrass depends highly on soil texture (Bowman et al., 2002), N-source (Guillard
and Kopp, 2004), fertilization rate and timing (Mangiafico and Guillard, 2006), the amount and timing of irrigation/rainfall (Paulino-Paulino et al., 2008; Petrovic, 1990), and the season of application (Petrovic, 1990). The worst-case scenario for NO$_3$-N leaching is the application of a soluble N source at a rate higher than the recommended rate, to a sandy soil that is over irrigated. Although trends relating such factors to the amount of NO$_3$-N leaching observed in the field have been identified, there is still a lack of detailed understanding, and more research are needed to quantify the effect of N forms, timing and rate of irrigation and fertilization, and soil textures on N-leaching.

To determine the amount of NO$_3$-N leaching from plant root zones, NO$_3$-N concentration is often measured in leachate collected from suction lysimeters (Gross et al., 1990). This technique requires a great deal of replication to evaluate the multiple factors that may contribute to NO$_3$-N leaching. Therefore, determining the factors contributing to NO$_3$-N leaching in an extreme scenario with coarse soil, highly soluble fertilizer, high irrigation and frequency is difficult. Models that predict flow and transport processes in soils are increasingly being applied to address practical problems. The use of simulation models allows extrapolation in time and space of data from leaching experiments and monitoring studies (Vanderborght et al., 2005). In recent years, many researchers have used model-based simulation methods to quantitatively evaluate water drainage at the farm level, while many non-point source (NPS) contaminant transport models have been developed to assess chemical transport over a wide range of topographies, soil types, climatic conditions, and management practices. For instance, deJong and Bootsma (1997) used the Soil Water Actual Transpiration Rate Extended (SWATRE) model to estimate water deficits and surpluses during the growing season in Ontario, Canada to estimate water drainage
from the root region and crop water deficiency. Heng et al. (2001) applied a water balance model to analyze multi-year water drainage of a pasture in Australia. Buchleiter et al. (1995) simulated the effects of over irrigation by 40% on crop yield, percolate produced, and NO$_3$-N leaching after the Root Zone Water Quality Model (RZWQM) was calibrated for a center pivot irrigated corn system. Sogbedji (2001) calibrated the Leaching Estimation and Chemistry Model (LEACHM) model to simulate N fate and transport under variable cropping histories and fertilizer rates in alfalfa. However, each model was developed for a specific use and has its own limitations and is not applicable to turfgrass systems due to specific input requirements. For example, the primary use of RZWQM is as a tool for assessing the environmental impact of alternative agricultural management strategies on the subsurface environment including conservation plans on field-by-field bases, tillage and residue practices, crop rotations, and planting dates and densities. The required information for LEACHM includes planting, emergence, maturity and harvest dates. In each case, these models are not suitable for simulating water and solute transport in turfgrass systems because turfgrass management is fundamentally different from agronomic crop management. Most turfgrasses are perennial while most agronomic crops are seeded and harvested during a single growing season. In addition, turf has a relatively consistent growth rate across the growing season, resulting in different water and N requirements from agronomic crops. Turfgrasses also require regular mowing, and the clippings can supply N consistently if they are returned back to the turf area. Some fertilizers developed specifically for turfgrasses may also have different N release rates than commonly used agronomic crop fertilizers. Because of these differences in fertilization and maintenance, the solute boundary conditions for
turfgrasses are very different from agronomic crops, and very few water or N simulations for turfgrasses have been reported.

Numerical model Hydrus-1D is a public domain computer software package that simulates the one-dimensional movement of water, heat and multiple solutes in variably saturated media (Simunek et al., 2008). The program allows analysis of water and nutrient flow and transport in unsaturated, partially saturated, or fully saturated media. Hydrus-1D draws on the Richards equation for simulation of soil water dynamics. To parameterize the Richards equation, Hydrus-1D uses, among others, the modified Mualem-van-Genuchten model (Vogel and Cislerova, 1988) to describe soil water retention and soil hydraulic conductivity. The model is widely accepted in both the research and the engineering communities and has been extensively verified by Simunek et al. (2008) by comparing model results with available analytical solutions for solute transport and with other numerical models for water flow. Hydrus-1D has been successfully applied in numerous studies to simulate and quantify improved management strategies and update irrigation standards for cotton (Forkutsa et al., 2009), to evaluate the leaching risks of N-Nitrosodimethylamine (NDMA) under fields irrigated with reclaimed wastewater (Haruta et al., 2008), to study ground water movement into the root zone and the uptake of ground water in a 10-year-old Populus euphratica woodland (Zhu et al., 2009), as well as to investigate the threat of heavy metal contamination to subsoil and ground-water quality (Ngoc et al., 2009). However, Hydrus-1D has not been applied to turfgrass water and N transport systems.

Therefore, it was the objective of this research to utilize observed soil water content data, soil ammonium (NH₄-N) and nitrate (NO₃-N) data, and boundary condition assumptions for turfgrasses to calibrate the water and N transport process in the Hydrus-1D model, and to verify the model among different turf species.
Subsequent objectives included the application of the calibrated and verified Hydrus-1D model to different turfgrass management scenarios (overirrigation, overfertilization, different soil textures) incorporating the potential factors that may affect N leaching and estimation of potential water costs and N-leaching under different management scenarios in Intermountain West region (IMW).

MATERIALS AND METHODS

A detailed description of the research materials and methods is provided in chapter 3. The methodology for determining soil water and N content in the turfgrass areas and the input of Hydrus-1D model is described here.

1. Experimental Site

The field experiment was conducted at the Utah Botanical Center (UBC), Kaysville, UT, U.S.A. The test site has a high mountain desert climate, with temperature extremes ranging from -30°C in January to 41°C in July. Average daily temperatures range from -4°C in January to 24°C in July. Soil at the test site is a Kidman fine sandy loam (coarse-loamy, mixed, mesic Calcic Haploxeroll) (USDA, 1968).

Nine drainage lysimeters were divided into woody plant, turfgrass and perennial hydrozones. Three replicates of Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG) were planted in the turf hydrozone of each lysimeter. Four Acclima® soil moisture sensors (Acclima Inc. Meridian, ID, USA) were installed in each turf hydrozone at depths of 80, 45, 20 and 5 cm, and measured volumetric soil
water content representing soil layers between 100-60, 60-30, 30-10, and 10-0 cm every hour, respectively.

Irrigation was controlled by an Acclima\textsuperscript{®} CS 3500 controller based on soil moisture sensors placed at 5 cm in a “master” hydrozone for each turf species. Irrigation water was distributed with 15.24 cm pop-up sprinklers (Rain Bird\textsuperscript{®} 1800 Series) with 1.83 m variable arc nozzles (Rain Bird\textsuperscript{TM} 6VAN). The volume of irrigation was recorded by DLJ\textsuperscript{®} 1.91 cm flow meters (Daniel L. Jerman Co., Hackensack, NJ, USA) connected to a Campbell Scientific\textsuperscript{®} CR1000 data logger (Campbell Scientific, Logan, UT, USA) and the depth of irrigation was obtained by dividing the total water volume applied by the turf area.

Precipitation data was recorded by a TR-525i tipping bucket rain gauge (Campbell Scientific, Logan, UT, USA) next to the plots and a reference evapotranspiration (ETo) dataset was obtained from the UBC weather station. Leachate from each landscape drained to collection wells adjacent to the plots, and was quantified using dipper trays connected to a CR1000 data logger (Campbell Scientific, Logan, UT, USA).

Spring and summer fertilizer composition and fertilization rates and timing for each turfgrass species are shown in Table 4-1. Soil samples from each turfgrass area were collected at depths of 0-10, 10-20, 20-30, 30-40, 40-60, and 60-80 cm using a soil auger on June 4, July 6, August 10 and September 7, 2010, respectively. Three cores per plot were collected and the samples in the same soil layers were uniformly mixed together and stored under refrigeration. Soil samples were ground for analysis at the Utah State University Analytical Laboratory (USUAL). Subsamples of 5 g each from every sample were taken and soil N was extracted with 20 ml of 2mol L\textsuperscript{-1} KCl. The NO\textsubscript{3}-N and NH\textsubscript{4}-N concentrations of the filtered soil solutions were analyzed
using Lachat’s QuikChem® 8500 Series 2 Flow Injection Analysis System (Lachat Instruments, Loveland, CO, USA).

Measurements of the effective turfgrass root length distribution were taken in May 2010. The sampling method was the same as that for soil N content analysis, and soil was washed from the roots in the lab. Root length density was measured by a modified line intersect method (Tennant, 1975). Roots were cut into 1 cm lengths and randomly placed into transparent dish which was divided into 1×1 cm squares. The number of intersections of roots and both vertical and horizontal lines were counted.

Root length (R) = number of intercept (N) * length conversion factor. \[\text{(1)}\]

where the conversion factor for 1 cm grid squares is 0.7857.

Root length density = root length / soil volume \[\text{(2)}\]

Root distribution determined the relative intensity of the potential root water uptake distribution. Measured root length density distributions between 0-80 cm were used to determine the relative density of root distribution for the three turf species, a required input of the Hydrus-1D model (Table 4-2).

2. Soil Properties

The topsoil and subsoil van Genuchten water retention curve (van Genuchten, 1980) property parameters [\(\theta_r\), \(\theta_s\), \(\alpha\) and \(n\), and the saturated hydraulic conductivity (Ks)] were estimated by Retention Curve (RETC) neutral network prediction (van Genuchten et al., 1991) based on the soil texture from the USDA soil survey (coarse-loamy, mixed, mesic Calcic Haploxeroll) (USDA, 1968) (Table 4-3). The optimization algorithm was used to fine-tune soil hydraulic parameters (\(\theta_s\), \(\alpha\), and \(n\)) until the simulated soil water contents agreed with observed soil water contents. In the survey soil profiles were divided into several layers, however, in this simulation, soil
profiles were divided into topsoil (30 cm depth) and subsoil based on visual observation since the soils were treated as topsoil and subsoil when the drainage lysimeters were constructed. In addition, soil textures determined at 0-10 cm and 30-50 cm in the survey were used for the topsoil and subsoil, respectively.

With the RETC estimated van Genuchten (VG) soil parameters as initial estimated parameters, the inverse module was used to optimize θ_s, α, and n. Daily average observed soil water content data at 80, 45, 20, and 5 cm for each turf species were used as an inverse data set. The VG parameters were optimized for each turf species, and the three sets of parameters were obtained for both topsoil and subsoil. Each set of parameters was also applied to the other two turfgrass species. The final parameters resulted from KBG inverse simulations that resulted in the least sum of squares of the residuals for all turf species (Table 4-3). The pore-connectivity parameter (l) was estimated by Mualem (1976) to be 0.5 as an average for many soils. However, Schaap and Leij (2000) recently recommended using -1 as an appropriate value for most soil textures, so -1 is used in this simulation (Table 4-3).

3. Theory Background

In Hydrus-1D, the solute transport equations consider advective-dispersive transport in the liquid phase, as well as diffusion in the gaseous phase. Two first-order degradation reactions are also included, one of which is independent of other solutes, and one of which provides the coupling between solutes involved in sequential first-order decay reactions. Hydrus-1D simulates N transport based on the chain reaction of N cycle. Simulation of the N dynamics in liquid, solid and gaseous phases are possible, however, only liquid and solid phases were considered in this simulation.
where \( c \) and \( s \) represent concentrations in the liquid and solid phases, respectively (Simunek et al., 2008). The terms \( c_1, c_2, \) and \( c_3 \) are the concentrations of NH\(_4\)-N, NO\(_2\)-N and NO\(_3\)-N in the liquid phase, respectively, and \( s_1 \) is the concentration of NH\(_4\)-N in the solid phase. Also, \( k_{c1}, k_{s1}, k_{c2}, \) and \( k_{c3} \) are the transformation parameters and represent the rate constants of the different processes in the liquid and solid phases.

Ammonium has a positive charge and can be absorbed by negatively charged soil particles, and adsorption can be subdivided into instantaneous (equilibrium) adsorption and kinetic (or non-equilibrium) adsorption. For the simulation, NH\(_4\)-N was assumed to have linear equilibrium adsorption described as

\[
s = K_d c, \tag{3}
\]

where \( K_d \) is the distribution coefficient [L\(^3\)M\(^{-1}\)] or the slope of the adsorption isotherm. Total concentration consisted of a dissolved solute concentration \( c \) and adsorbed concentration \( s \),

\[
C_T = \theta c + \rho_b s \tag{4}
\]

where the dimension for \( s \) is mass of solute adsorbed per mass of soil (MM\(^{-1}\)), and \( \rho_b \) is soil bulk density [ML\(^{-3}\)].

4. Hydrus-1D Model Construction and Inputs

Water movement and N transport were simulated using the Hydrus-1D numerical model (Version 4.14).

4.1. General Information

The lysimeter soil profile depth ranged from 1.2 to 1.4 m, and since the total depth of soil dictates the hydraulic potential at the outlet, the simulated soil profile
was 1.4 m. Topsoil and subsoil depths were 30 and 110 cm in the simulation, respectively. The simulation period began on May 1 and ended on Sep. 30 for hydraulic transport simulations, and began on June 4 and ended on Sep 30th for N transport simulations since the initial N contents were obtained on June 4. Time discretization was broken down as: initial time step $1 \times 10^{-3}$ day, minimum time step $1 \times 10^{-3}$ day, and maximum time step 0.1 day.

4.2. Root Water Uptake Parameters

For the determination of root water uptake, the method proposed by Feddes et al. (1978) and modified by van Genuchten (1987) to include multiplicative water and osmotic stress was applied. The inherent water stress reduction term was parameterized with the function proposed by Feddes et al. (1978). The suggested Hydrus-1D database for turfgrass root water uptake parameters included: $h_1 = -10$ hPa, $h_2 = -25$ hPa, $h_{3\text{high}} = -240$ hPa, $h_{3\text{low}} = -360$ hPa, and $h_4 = -8,000$ hPa.

4.3 Boundary and Initial Conditions

The upper boundary condition (BC) was an atmospheric BC with a surface layer, in which the measured daily rainfall and potential ET for the entire simulation period were used as a time-variable boundary for the soil surface. Applied irrigation depths were added to rainfall data. Potential evaporation and transpiration were entered as separate inputs to time steps, which could be per day, hour, or minute, and daily time steps were used for the simulation. However, the irrigation/precipitation timing in a day was specified as minutes for irrigation and hours for precipitation. Evaporation from soil was neglected as the site was covered by turfgrass and no bare soil existed in the turf areas, except for BG in May. Buffalograss, a warm-season grass, began
actively growing in early June 2010, and the evaporation was set as \( \frac{1}{2} \) ET0 since soil was covered by dry turf, and the corresponding potential evaporation was set as zero. 120% of ET0 was entered as potential transpiration for TF since TF had higher actual ET in 2010 (chapter 3). In correspondence with field conditions, “ponding” i.e., water building up on the soil surface, was allowed to take place. Since the plots were constructed as lysimeters, no free drainage was allowed and a seepage face at -1.4 m with \( h=0 \) was chosen as the lower BC.

4.4 Nitrogen Upper Boundary Conditions

Solutes were introduced to the model domain through the amount and concentration of rainfall/irrigation water on the day of application. Initial concentrations of the soil solutes were set as the averaged measurement of the three plots of each turf species on June 4, 2010. There were two N sources in the simulation, fertilizer application and clipping decomposition. The fertilizer was composed of urea, \( \text{NH}_4 \)-N, \( \text{NO}_3 \)-N, and coated water insoluble N (WIN). Since only \( \text{NH}_4 \)-N, \( \text{NO}_2 \)-N and \( \text{NO}_3 \)-N were considered in this simulation, urea was considered as \( \text{NH}_4 \)-N in the simulation, and \( \text{NH}_4 \)-N and \( \text{NO}_3 \)-N were treated as fast-release fertilizers. The flux concentrations for \( \text{NH}_4 \)-N and \( \text{NO}_3 \)-N were obtained by dividing total N by the total volume of irrigation on the day fertilizer was applied. Water insoluble N was assumed to function as \( \text{NO}_3 \)-N and to be released at a fixed rate during growing season. The flux concentration of coated slow release \( \text{NO}_3 \)-N was the total amount of \( \text{NO}_3 \)-N in WIN particles divided by the total irrigation amount during the growing season. For returned clippings, since N is built into proteins and other complex molecules in plants, organic matter is decomposed and \( \text{NH}_4 \)-N is released through mineralization (Troeh and Thompson, 1993), a consistent decomposition rate for \( \text{NH}_4 \)-N during
The growing season was assumed. It was also assumed that 90% of N of clippings was released back to the plots in the form of NH$_4$-N (Kopp and Guillard, 2002), and that NH$_4$-N was released with every irrigation/precipitation event. The N contents of clippings for each species were estimated based on mowing frequency, fresh and dry weight of clippings, and N content of clippings. The KBG and TF plots were mowed bi-weekly, while BG was mowed once every three weeks in 2010, and fresh weights of 90 g m$^{-2}$ clippings were estimated for each mowing. The N content of KBG, TF and BG clippings were 3.1%, 3.6%, and 3.5% of dry weight (Hallock et al., 1965; Hull, 1992) respectively, and 27%, 30%, and 35% of dry weight out of fresh weight for KBG, TF, and BG were estimated, respectively. The NH$_4$-N flux BCs were estimated based on the previous assumptions.

4.5 Solute Transport and Reaction Parameters

Three solute chain reactions among NH$_4$-N, NO$_2$-N and NO$_3$-N were simulated. Solute transport and transformation parameters for each solute were converted from the Hydrus-1D built-in example “TEST 3 - solute transport and nitrification chain”. Ammonium was set as passive uptake while NO$_3$-N was set as both passive and active uptake, depending on maximum allowed concentration for passive root solute uptake (C$_{max}$) (Simunek and Hopmans, 2009). Since the public release of Hydrus-1D allows active uptake for one solute simulation only, the internal version of active uptake code for NO$_3$-N was obtained from Dr. Jiri Simunek (personal communication, 2011).

5. Model Performance Criteria

Correlations between simulated and observed data were developed. Correlation coefficient ($r^2$) is a quantitative criterion of the goodness of fit of the model and
reflects similar and/or dissimilar trends between observed and simulated data. The closer the correlation is to 1, the more accurate the model is. In addition, two objective functions were calculated for N transport and transformation simulation to evaluate predicted vs. measured ammonia and nitrate data. The first object function is the root mean square error (RMSE), which is calculated in the following manner:

$$RMSE = \left[ \frac{\sum_{i=1}^{N}(P_i - O_i)^2}{N} \right]^{1/2}$$  \hspace{1cm} (5)

where $P_i$ and $O_i$ are the $i$th predicted and observed values of interest, respectively. The values of RMSE is in the same units as the corresponding data, and is a measure of the average deviation of the predicted data that observed. The second objective function is Willmott’s index of agreement ($d$), expressed as

$$d = 1 - \left[ \frac{\sum_{i=1}^{N}(P_i - O_i)^2}{\sum_{i=1}^{N}||P_i'||||O_i'||} \right]$$  \hspace{1cm} (6)

where $P_i' = P_i - O_m$, $O_i' = O_i - O_m$, and $O_m$ is the mean observed value (Willmott, 1982). The value of $d$ is an index of how well the predicted and observed deviations about $O_m$ correspond to each other, both in magnitude and sign. It varies between 1.0 and 0.0, with 1.0 representing perfect agreement and 0.0 representing one of many forms of total disagreement. The two objective functions (RMSE and the $d$ index) in conjunction quantify the agreement between simulated and observed results.

6. Scenario Simulations

During the course of the study, the research plots were managed for efficient irrigation applications based on soil moisture and best fertilization practices based on Utah State University Cooperative Extension recommendations, so negligible amounts of leachate were collected during 2010 growing season for each plot. However, homeowners and other lawn managers may over apply irrigation and
fertilizer, which may result in NO$_3$-N leaching. Therefore, after the model was calibrated with observed soil water content data, water leaching data, soil NH$_4$-N and NO$_3$-N content data for KBG, and verified with data for TF and BG, the model was applied for the three species under scenarios of over irrigation, over fertilization, and different soil textures. The NO$_3$-N leaching potential under extreme irrigation and fertilization scenarios were simulated. The scenarios included: (1) monthly fertilizer application (2× fertilizer); (2) 150% optimized irrigation (1.5× irrigation), (3) 150% optimized irrigation and monthly fertilizer application (1.5× irrigation + 2× fertilizer); (4) 200% optimized irrigation (2× irrigation); (5) 200% optimized irrigation and monthly fertilizer application (2× irrigation + 2× fertilizer); (6) sandy loam soil; (7) loam soil; (8) clay loam soil. The irrigation and fertilizer application timing and rate applied in the research plots for each turf species, and the soil in the research plot were taken as controls to compare to the simulation results in irrigation, fertilization and soil texture scenarios. In the scenario simulations, two assumptions were made: (1) under different irrigation, fertilization and soil conditions, the solution transport and transformation parameters were the same; (2) the released NH$_4$-N amount from clippings were the same for each scenario.

6.1. Monthly Fertilization

Fertilizer (33-0-0) was applied four times during the simulation period (48.8 kg ha$^{-1}$ N). At the beginning of each month, urea, NH$_4$-N and NO$_3$-N were applied with the irrigation on the day fertilizer was applied, and WIN were applied with each irrigation/precipitation event.
6.2. 1.5× and 2× irrigation

According to irrigation surveys conducted in Salt Lake City (SLC) in recent years, 150% to 200% of recommended irrigation amounts are typically applied to turfgrass areas. Homeowners may apply water without an understanding of the water requirements of turf, or set up their irrigation timers or controllers according to the water requirements of turf in midsummer and never change the settings, resulting in over irrigation during early and late growing season. For the simulation of these scenarios, the irrigation/precipitation rate was set to the 150% and 200% of the optimized irrigation scenario without changing the timing of irrigation/precipitation. And the NH$_4$-N and NO$_3$-N concentration during each irrigation/precipitation event was decreased accordingly to make sure the total N application rate was the same as that of corresponding fertilization scenarios.

6.3. Soil Texture Scenarios

Sandy loam, loam and clay loam soil textures were chosen to determine the effect of soil texture on N leaching potential. The soil property VG parameters were obtained from the Hydrus-1D built-in database (Table 4-4). All the input parameters and variable BCs, and initial BCs were the same as that of the control except for the soil property parameters.

RESULTS AND DISCUSSION

1. Water Transport Calibration and Verification

Model calibration consisted of iteratively adjusting the soil hydraulic parameters, so that simulated soil water contents agreed with measured soil water contents to an
acceptable accuracy. Because model calibration, or parameter optimization, was an indirect approach for estimating soil hydraulic parameters from soil water transport data, independent data were used to validate the calibrated soil hydraulic parameters. In this research, the KBG data set was used to calibrate the model and to estimate all necessary parameters, while the TF and BG (validation or verification) data sets served to compare predicted and measured data values using the parameters calibrated against the KBG data set. Thus, if simulated soil water contents were in acceptable agreement with measured soil water contents for TF and BG data set, the model was considered to be validated for given conditions. Once validated, the model could be used to simulate non-measured conditions.

Simulated and observed soil $\theta$ values at 5, 20, 45, and 80 cm for three turf species were developed (Fig. 4-1). Simulated and observed water contents followed a similar trend without much difference, indicating that the model was able to simulate time varying boundary flux, and suggesting a good fit between the simulated water contents and observed water content at different depths for KBG, TF and BG. Correlations were also developed for simulated and observed $\theta$’s at 5, 20, 45, and 80 cm for the three turf species (Fig. 4-2). With the exception of the 5 cm simulation in TF, all correlations were between 0.73 and 0.89, indicating a very good fit for the model simulations, and that Hydrus-1D can be used to simulate the water distribution with acceptable accuracy. On May 20-29, 2010, $\theta$ peaks occurred at 45 cm and 80 cm for all three turf species, though these peaks were not simulated by the Hydrus-1D model. These peaks may have resulted from increased $\theta$ in the subsoil resulting from storms during that period and relatively slow drainage of the lysimeters (Fig. 4-1, Fig. 4-3). Overall, the Hydrus-1D model simulated water transport very well. Simulated $\theta$ was overestimated at 80 cm for the three turf species when compared to observed $\theta$’s
late in the growing season, especially for TF. This may have resulted from inaccurate VG hydraulic soil parameters for deeper soils. According to the soil survey, the soil profiles were divided into five layers in the top 1m according to their textures, however, only two soil layers were assumed in this simulation, and the inverse simulation for subsoil was based on the soil texture at the 30-50 cm depth.

Simulated and observed leachate depths for KBG, TF and BG during May and September were developed (Fig. 4-4) and agreed well for each turf species, suggesting a good fit of the water transport simulation with Hydrus-1D. All three turf species had significant amounts of leachate at the beginning of growing season, while TF had the greatest amount of leachate. The greater amount of drainage in early season for three turf species may have resulted from the storms that occurred May 20-29, which were coincident to the $\theta$ peaks at 45 and 80 cm deep soil layer in late May (Fig. 4-1). In the late growing season, all three turf species had no drainage for either the simulation or the observation.

2. N Concentration Calibration and Verification

With the inversed simulated soil VG parameters for topsoil and subsoil, three N form chain reactions were simulated. The N simulation period began on June 4th since the original N contents were obtained that day. Observed NH$_4$-N content for 0-10, 10-20, 20-30, 30-40, 40-60 and 60-80 cm soil layers and simulated soil NH$_4$-N contents at observation nodes of 5, 15, 25, 35, 50, and 70 cm were plotted over time from June to September for three turf species (Fig. 4-5). Observed and simulated soil NH$_4$-N (ppm) contents from 0 to 80 cm for the four observation dates and the three turf species were plotted (Fig. 4-6). The simulated NH$_4$-N (ppm) contents fit the measured data very well for KBG, TF and BG. In the soil profile plot for each observation day,
the simulation fit the measured data very well. Most of the simulations were within the range of one standard error of the averaged measured data and correlations ranging from 0.85 to 0.92 for simulated and observed NH$_4$-N contents for each turf species were determined (Fig. 4-7). In addition, simulated soil NH$_4$-N contents at observation nodes of 5, 15, 25, 35, 50, 70 and corresponding soil layers NH$_4$-N contents were plotted according to depth integrating three turf species, and correlation ranged from 0.68 to 0.87 for different soil layers (Fig. 4-8). Statistical means, standard deviation (SD) for observed NH$_4$-N content, and RMSE and Willmott’s $d$ index for observed and predicted NH$_4$-N data for each observation depth and species are shown in Table 4-5, and the $d$ index ranged between 0.85 and 0.95 for all depth and species except 10-20 cm soil layer, indicating a very good agreement between simulated and observed NH$_4$-N content. Above results indicate that the Hydrus-1D model was able to simulate NH$_4$-N changes in the soil profile with the selected NH$_4$-N transformation parameters, and this agreement between observation and simulation indicates that the boundary condition assumptions for N release from clippings worked well for the turfgrasses studied.

For NO$_3$-N content, however, the simulation did not fit the measured data as well as NH$_4$-N. Simulated NO$_3$-N content was generally higher than measured NO$_3$-N content between 10 to 40 cm in the soil profile (Fig. 4-9 and Fig. 4-10), and correlations ranging from 0.18 to 0.47 for simulated and observed soil NO$_3$-N content were determined (Fig. 4-11). These results did not indicate good agreement between simulated and observed NO$_3$-N values and may be a result of NO$_3$-N being a late product in the simulation, allowing many transport and transformation parameters to affect the NO$_3$-N content in soil profile, and meaning that the nitrogen transport and transform parameters need to be sharpened. Simulated soil NO$_3$-N contents at
observation nodes of 5, 15, 25, 35, 50, 70 and corresponding soil layers NO$_3$-N contents were plotted according to depth integrating three turf species as well (Fig. 12). The results indicated that the model can simulate NO$_3$-N contents in top 10 cm and in deep soils of 40-60 and 60-80 cm, with the correlations of 0.67, 0.42, and 0.75, respectively (Fig. 12). Similar results were obtained in Willmott’s $d$ index and RMSE for simulated and observed NO$_3$-N content data (Table 4-5). The good agreement between simulated and observed NO$_3$-N contents in the top 10 cm soil suggests a reasonable N boundary condition and nitrogen release rate for slow release fertilizer were specified, and the good agreement in deep 60-80 cm soil indicates the nitrate leaching rate obtained from this simulation has some merit in scenario simulations. The over-predicted NO$_3$-N contents in the soil layers between 10 and 40 cm may be resulted from the compromised NO$_3$-N denitrification rate parameter in sub soil, suggesting different soil NO$_3$-N denitrification rates according to soil depth are needed in the Hydrus-1D model, since denitrification rate is the most influential parameter on NO$_3$-N leaching (Almasri and Kaluarachchi, 2004).

Although simulated NO$_3$-N content did not fit observed NO$_3$-N content in 20-40 cm soil layer, and may have resulted in over or underestimated NO$_3$-N leaching at the bottom of root zone, the trends of NO$_3$-N leaching under different scenarios should be the same. Following calibration and validation, the water transport parameters and N chain reaction parameters obtained from the calibration and verification process were used to simulate the different irrigation and fertilization scenarios to compare N leaching from turfgrass root zone, and simulate three soil texture scenarios to demonstrate the influence of soil texture on N leaching. Since turf roots reached a depth of 80 cm, N leaching below that depth would not be available to turf. An 80 cm deep soil profile was used in scenario simulations. In a field condition, there is no
barrier to prevent water from moving down freely, therefore, free drainage boundary conditions were used in scenario simulations; all other parameters are the same as that in the calibration and verification process.

3. Scenario Simulations

The scenario simulations clearly indicated that over irrigation and over fertilization can increase N leaching significantly for KBG, TF and BG (Fig. 4-13). However, 2× fertilization alone without over irrigation generated the same N leaching as the control in three turf species, indicating that extra water is the main controlling factor in N leaching (Fig. 4-13). Similar results were obtained in that the total amount of NO$_3$-N leaching did not differ significantly for the two fertilizer application rates (25 and 50 kg ha$^{-1}$ N) if irrigation was managed very well (Wong et al., 1998). Under the 1.5× irrigation and 2× irrigation scenarios, 2× fertilizer application caused more NO$_3$-N leaching when compared to over irrigation alone. When fertilizer application scenarios were held constant, increasing application of water caused increased NO$_3$-N leaching. However, when irrigation scenarios were held constant, the NO$_3$-N leaching only occurred under the over irrigation scenario (Fig. 4-13). The most extreme scenario (2× irrigation + 2× fertilizer) resulted in the most NO$_3$-N leaching (61, 77, 40 kg ha$^{-1}$ N for KBG, TF, and BG, respectively). The 2× irrigation scenario resulted in more NO$_3$-N leaching than the 1.5× irrigation + 2× fertilizer scenario for all three turf species, confirming that irrigation is the greater contributing factor to NO$_3$-N leaching. Under the control scenario, TF had the least NO$_3$-N leaching while BG had the greatest (1.43 kg ha$^{-1}$ N vs. 4.48 kg ha$^{-1}$ N). However, under over irrigation scenarios, TF had the greatest NO$_3$-N leaching while BG had the least. This was because TF had
the greatest irrigation in the control plot, which resulted in more extra water under
over irrigation scenarios.

The effect of soil texture on NO₃-N leaching was simulated as well (Fig. 4-14),
and the results suggested that sandy loam soils facilitated NO₃-N leaching and that
clay loam soils mitigated NO₃-N leaching for all turf species. The leaching potential
of N in clay loam soil was almost negligible, while that in sandy loam was about three
times that of the control for all turf species (Fig. 4-14). This confirms that more
permeable soils are prone to leaching compared to the less permeable soils, similar
results were demonstrated by Ajdary et al. (2007).

For all three turf species, NO₃-N leaching occurred before July and was
concurrent with water drainage (Figs 4-4 and 4-14), further indicating that NO₃-N is
transported with water and that water is the main contributing factor to NO₃-N
leaching in turf systems. Both clay loam and loam soils decreased NO₃-N leaching
when compared to the control soil (Kidman fine sandy loam), and clay loam soils
decreased NO₃-N leaching to almost zero for all three turf species. The Kᵢ value for
sandy loam, loam, and clay loam soils were 1061, 249.6, and 62.4 mm day⁻¹,
respectively. Since sandy loam soils had the least capacity to retain water, NO₃-N was
leached downward more quickly under the management scenarios.

In all three soil texture scenario simulations, BG had the greatest NO₃-N leaching
while TF had the least. This may be because of higher NO₃-N uptake by TF and lower
NO₃-N uptake by BG. The model allowed roots to uptake NO₃-N passively, so root
uptake of NO₃-N increased with root uptake of water. As a result, TF had the least
NO₃-N leaching while BG had the greatest NO₃-N leaching.
4. Uncertainties

Although the calibrated Hydrus 1-D model simulated water transport and NH$_4$-N content consistent with measurements, simulated NO$_3$-N was less well correlated to measured subsurface concentrations. There is, therefore, a level of uncertainty as to the range of applicability of the model for different management scenarios. For instance, with different soil textures, only soil hydraulic parameters changed in the scenario simulation, nitrification and denitrification parameters were kept the same for the different soils in the soil texture scenario simulations in this study. However, it has been demonstrated that soil type can affect N transformation rates (Shearman, 1986), and the microbacteria groups, K$_d$ values for NH$_4^+$ adsorption, and root distribution may all change with soil texture, which would affect N transformation and transport parameters and ultimately affect NO$_3$-N leaching. In addition, N transformation rates were kept the same for the entire simulation period under all turf species and irrigation schedules. In reality, nitrification slows or stops when the soil is too cold, too hot, too dry, or is deficient in oxygen, while denitrification favors waterlogged conditions, suggesting that parameters for nitrification and denitrification may be different under different irrigation schedules from season to season, and may be different for different turf species. Furthermore, with increased fertilizer application, plant uptake of N may increase resulting in higher N contents and greater N amounts in clippings. If these clippings are returned to the turf area as showed in this study, the amount of N being returned to the area may also be increased. Although uncertainties exist under scenario simulations, the trend of N transformation and transport should be the same, so the simulation results are still be qualitatively meaningful.
5. Implications

The Intermountain West region has faced severe water shortages in recent years, and most water waste is caused by improper or inefficient landscape irrigation. As a result, water prices in IMW are higher either in summer or with higher tiers of water use to encourage homeowners to conserve water when irrigating their landscapes. According to the highest 2011 water prices in Denver, Las Vegas, Phoenix, and Salt Lake City, the potential extra water cost for a family with 185.8 m² KBG residential landscape in the above mentioned cities was estimated (Table 4-6), and potential N-leaching under different irrigation and fertilization conditions were determined according to simulation results. In Denver, which has the highest water price among the four cities in IMW, $380 may be spent to maintain a well-watered KBG turf, while $570 may be spent for 1.5× irrigation and $780 for 2× irrigation. In addition, over-irrigation could result in from 743 (1.5× irrigation) to 1133g (2× irrigation) N leaching to ground water if the turf were fertilized monthly at a rate of 48.8 kg N/ha. Under recommended irrigation and fertilization schedules, 12% of applied N could be leached to ground water, while under 1.5× irrigation and 2× fertilizer, and 2× irrigation and 2× fertilizer scenarios, 41% and 62% of applied N could be leaching to ground water.

CONCLUSIONS

Results presented in the paper described the calibration and verification process of water and N transport processes through Kentucky bluegrass, tall fescue and buffalograss turfgrass systems. Calibration and validation results showed that Hydrus-1D may be used for simulation of water and N distribution and leaching. Scenario
simulation results revealed that over irrigation is the main contributing factor to NO$_3$-N leaching in turfgrass systems, and that NO$_3$-N leaching increased as irrigation increased. Over fertilization alone, with optimized irrigation, did not increase NO$_3$-N leaching during the growing season. However, under conditions of over irrigation, over fertilization NO$_3$-N leaching increased significantly. Soil textures had a significant effect on N leaching as well. In coarse-textured, sandy loam soils, more N can be leached down than in loam and clay loam soils under the same irrigation and fertilization conditions. Irrigation amounts of 1.5× and 2× combined with 2× fertilizer could result in $570 to $780 in additional water costs for a 185.8 m$^2$ KBG turf in Denver, as well as result in between 743 and 1133 g N leaching to ground water. Therefore, from the scenario simulation results, homeowners and turfgrass managers should pay the most attention to irrigation schedules and fertilizer application, especially on coarse textured soils (sandy loam and sand), to decrease the potential for NO$_3$-N leaching to ground water.

REFERENCES


Table 4-1. Fertilizer composition and formula, fertilization rate and timing for Kentucky bluegrass (KBG), tall fescue (TF) and buffalograss (BG) (2010 growing season).

<table>
<thead>
<tr>
<th>Application date</th>
<th>Turf fertilized</th>
<th>Fertilizer components</th>
<th>Application N rate kg ha(^{-1})</th>
<th>Fertilizer formula Ammonium %</th>
<th>Urea %</th>
<th>Nitrate %</th>
<th>WIN %</th>
</tr>
</thead>
<tbody>
<tr>
<td>June 4, 2010</td>
<td>KBG, TF, BG</td>
<td>18-9-18</td>
<td>48.8</td>
<td>1.90</td>
<td>6.90</td>
<td>5.40</td>
<td>3.80</td>
</tr>
<tr>
<td>July 27, 2010</td>
<td>BG</td>
<td>33-0-0</td>
<td>48.8</td>
<td>2.22</td>
<td>3.93</td>
<td>8.53</td>
<td>18.32</td>
</tr>
<tr>
<td>August 27, 2010</td>
<td>KBG, TF</td>
<td>30-0-0</td>
<td>48.8</td>
<td>2.31</td>
<td>3.50</td>
<td>7.43</td>
<td>16.76</td>
</tr>
</tbody>
</table>
Table 4-2. Relative intensity of the potential root water uptake distribution for Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG). Numbers in parentheses indicate the absolute root length density of KBG, TF and BG in the top 10 cm soil layer (cm root/cm³ soil).

<table>
<thead>
<tr>
<th>Depth (cm)</th>
<th>KBG</th>
<th>TF</th>
<th>BG</th>
</tr>
</thead>
<tbody>
<tr>
<td>0-10</td>
<td>1 (69)</td>
<td>1 (100)</td>
<td>1 (7)</td>
</tr>
<tr>
<td>10-20</td>
<td>0.1</td>
<td>0.15</td>
<td>0.8</td>
</tr>
<tr>
<td>20-30</td>
<td>0.1</td>
<td>0.1</td>
<td>0.6</td>
</tr>
<tr>
<td>30-40</td>
<td>0.05</td>
<td>0.05</td>
<td>0.4</td>
</tr>
<tr>
<td>40-50</td>
<td>0.03</td>
<td>0.04</td>
<td>0.4</td>
</tr>
<tr>
<td>50-60</td>
<td>0.03</td>
<td>0.04</td>
<td>0.4</td>
</tr>
<tr>
<td>60-70</td>
<td>0.01</td>
<td>0.02</td>
<td>0.3</td>
</tr>
<tr>
<td>70-80</td>
<td>0.01</td>
<td>0.02</td>
<td>0.3</td>
</tr>
</tbody>
</table>
Table 4-3. Soil textures, estimated van Genuchten (VG) parameters, and final inversed VG parameters for top soil and subsoil.

<table>
<thead>
<tr>
<th>Soil Texture</th>
<th>RETC Estimated Parameters</th>
<th>Inverse fitted parameters</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>sand silt clay bulk density</td>
<td>θr θs α n Ks</td>
</tr>
<tr>
<td>Top soil (0-10)</td>
<td>45.9 37.2 16.9 1.6</td>
<td>0.049 0.3534 0.00158 1.4042 92.4</td>
</tr>
<tr>
<td>Subsoil (30-51)</td>
<td>42.5 43.2 14.3 1.6</td>
<td>0.045 0.3432 0.00136 1.4335 99.8</td>
</tr>
</tbody>
</table>
Table 4-4. Soil van Genuchten (VG) parameters for sandy loam, loam, and clay loam soils in simulation scenarios.

<table>
<thead>
<tr>
<th></th>
<th>$\theta_r$</th>
<th>$\theta_s$</th>
<th>$\alpha$</th>
<th>n</th>
<th>$K_s$</th>
<th>$l$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sandy loam</td>
<td>0.065</td>
<td>0.41</td>
<td>0.0075</td>
<td>1.89</td>
<td>1061</td>
<td>0.5</td>
</tr>
<tr>
<td>Loam</td>
<td>0.078</td>
<td>0.43</td>
<td>0.0036</td>
<td>1.56</td>
<td>249.6</td>
<td>0.5</td>
</tr>
<tr>
<td>Clay loam</td>
<td>0.095</td>
<td>0.41</td>
<td>0.0019</td>
<td>1.31</td>
<td>62.4</td>
<td>0.5</td>
</tr>
</tbody>
</table>
Table 4-5. Statistics mean, standard deviation (SD) for observed ammonia and nitrate data, objective function results for observed and predicted ammonia and nitrate data for each observation depth and species.

<table>
<thead>
<tr>
<th></th>
<th>0-10cm</th>
<th>10-20cm</th>
<th>20-30cm</th>
<th>30-40cm</th>
<th>40-60cm</th>
<th>60-80cm</th>
<th>KBG</th>
<th>TF</th>
<th>BG</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ammonia Mean</td>
<td>8.33</td>
<td>5.04</td>
<td>4.82</td>
<td>4.20</td>
<td>2.92</td>
<td>2.64</td>
<td>4.65</td>
<td>4.69</td>
<td>4.62</td>
</tr>
<tr>
<td>SD</td>
<td>3.14</td>
<td>2.40</td>
<td>2.54</td>
<td>2.09</td>
<td>1.51</td>
<td>1.47</td>
<td>3.16</td>
<td>3.05</td>
<td>2.52</td>
</tr>
<tr>
<td>RMSE†</td>
<td>2.06</td>
<td>2.19</td>
<td>1.33</td>
<td>1.15</td>
<td>0.87</td>
<td>0.76</td>
<td>1.72</td>
<td>1.73</td>
<td>1.00</td>
</tr>
<tr>
<td>d index‡</td>
<td>0.85</td>
<td>0.74</td>
<td>0.92</td>
<td>0.92</td>
<td>0.91</td>
<td>0.93</td>
<td>0.89</td>
<td>0.87</td>
<td>0.95</td>
</tr>
<tr>
<td>Nitrate Mean</td>
<td>2.82</td>
<td>1.60</td>
<td>1.55</td>
<td>1.47</td>
<td>1.51</td>
<td>1.38</td>
<td>1.71</td>
<td>1.54</td>
<td>1.92</td>
</tr>
<tr>
<td>SD</td>
<td>0.73</td>
<td>0.25</td>
<td>0.23</td>
<td>0.20</td>
<td>0.17</td>
<td>0.17</td>
<td>0.61</td>
<td>0.32</td>
<td>0.76</td>
</tr>
<tr>
<td>RMSE</td>
<td>0.58</td>
<td>1.26</td>
<td>1.22</td>
<td>0.97</td>
<td>0.39</td>
<td>0.49</td>
<td>1.07</td>
<td>0.68</td>
<td>0.89</td>
</tr>
<tr>
<td>d index‡</td>
<td>0.78</td>
<td>0.20</td>
<td>0.15</td>
<td>0.17</td>
<td>0.51</td>
<td>0.48</td>
<td>0.52</td>
<td>0.41</td>
<td>0.64</td>
</tr>
</tbody>
</table>

† Root mean squared error.
‡ Willmott’s index of agreement.
Table 4-6. Potential water cost and N-leaching for Kentucky bluegrass in Intermountain West cities for a 185.8 m² turf yard.

<table>
<thead>
<tr>
<th>Location</th>
<th>Water price $/100 cf</th>
<th>Water price $/m³</th>
<th>Recommend 1.5x irrigation+ 2x fertilizer $</th>
<th>2x irrigation+ 2x fertilizer $</th>
</tr>
</thead>
<tbody>
<tr>
<td>Denver</td>
<td>9.64</td>
<td>3.40</td>
<td>380</td>
<td>570</td>
</tr>
<tr>
<td>Las Vegas</td>
<td>4.58</td>
<td>1.62</td>
<td>180</td>
<td>270</td>
</tr>
<tr>
<td>Phoenix</td>
<td>3.77</td>
<td>1.33</td>
<td>148</td>
<td>223</td>
</tr>
<tr>
<td>Salt Lake City</td>
<td>1.98</td>
<td>0.69</td>
<td>78</td>
<td>117</td>
</tr>
<tr>
<td>N-leaching (g)</td>
<td></td>
<td></td>
<td>111</td>
<td>743</td>
</tr>
<tr>
<td>Percent N-leaching</td>
<td></td>
<td></td>
<td>12%</td>
<td>41%</td>
</tr>
</tbody>
</table>
Figure 4-1. Simulated and observed soil volumetric water contents at 5, 20, 45, and 80 cm for Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG) (2010 growing season).
Figure 4-2. Simulated volumetric water content (m$^3$/m$^3$) plotted against observed volumetric water content (m$^3$/m$^3$) at 5, 20, 45, and 80 cm for Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG).
Figure 4-3. Irrigation and precipitation (mm) during 2010 simulation period for Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG).
Figure 4.4. Simulated and observed water leachate depth for Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG) turf from May to September, 2010.
Figure 4-5. Observed and simulated soil NH$_4$-N (ppm) contents at 5, 15, 25, 35, 50, and 70 cm from June to September (2010) for Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG). Error bars indicate one standard deviation from the mean.
Figure 4-6. Observed and simulated soil NH$_4$-N (ppm) contents from 0 to 80 cm soil depth on four observation dates for Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG). Error bars indicate one standard deviation from the mean.
Figure 4-7. Simulated soil NH$_4$-N (ppm) content plotted against observed soil NH$_4$-N (ppm) content for Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG).
Figure 4-8. Simulated soil NH$_4$-N (ppm) content plotted against observed soil NH$_4$-N (ppm) content at 5, 15, 25, 35, 50, and 70 cm depth for corresponding 0-10, 10-20, 20-30, 30-40, 40-60, and 60-80 cm soil layers integrating Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG).
Figure 4-9. Observed and simulated soil NO$_3$-N (ppm) content at 5, 15, 25, 35, 50, and 70 cm from June to September (2010) for Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG). Error bars indicate one standard deviation from the mean.
Figure 4-10. Observed and simulated soil NO$_3$-N (ppm) content from 0 to 80 cm soil depth on the four observation dates for Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG). Error bars indicate one standard deviation from the mean.
Figure 4-11. Simulated soil NO$_3$-N (ppm) content plotted against observed soil NO$_3$-N (ppm) content for Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG).
Figure 4-12. Simulated soil NO$_3$-N (ppm) content plotted against observed soil NO$_3$-N (ppm) content for 5, 15, 25, 35, 50, and 70 cm depth for corresponding 0-10, 10-20, 20-30, 30-40, 40-60, and 60-80 cm soil layers integrating Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG).
Figure 4-13. Simulated NO$_3$-N leaching losses from different irrigation and fertilization simulation scenarios applied to Kentucky bluegrass (KBG), tall fescue (TF), and buffalograss (BG) including a control, 2× fertilizer application, 1.5× of recommended irrigation volumes, 1.5× of recommended irrigation volumes and 2× fertilizer applications, 2× of recommended irrigation volumes, and 2× of recommended irrigation volumes and 2× fertilizer applications.
Figure 4-14. Simulated NO$_3$-N leaching losses under different soil texture scenarios including control, sandy loam, loam, and clay loam soils.
ABSTRACT: Nitrogen (N) fertilization of urban turf areas, and potential nitrate (NO$_3$-N) leaching, may pose a hazard to ground-water quality. This research utilized a Geographic Information System (GIS) approach to estimate NO$_3$-N leaching mass from urban turf areas based on a one-dimensional N leaching model and to classify the NO$_3$-N leaching risk in the Salt Lake Valley, Utah, USA, based on soil texture. The methodology integrated a calibrated and verified Hydrus-1D N model, soil textures and urban turf areas to predict NO$_3$-N leaching to groundwater. Thirty United States Geological Survey (USGS) residential wells were installed and sampled in 1999 for NO$_3$-N concentration analysis. A relationship between estimated NO$_3$-N leaching from urban landscapes and groundwater NO$_3$-N concentration was developed to determine the effect of soil texture and landscaped area on NO$_3$-N leaching from urban landscapes. The GIS approach was used to estimate the NO$_3$-N leaching risk to groundwater under efficient irrigation and fertilization scenarios and over irrigation and over fertilization scenarios. The results showed that soil texture played a role in NO$_3$-N leaching from urban landscapes to groundwater, and shallow groundwater was more susceptible to surface contamination compared to deep groundwater. The GIS technique identified areas where improved irrigation and fertilization management could reduce landscape NO$_3$-N leaching significantly, resulting in fewer NO$_3$-N leaching risk areas in the Salt Lake Valley, Utah, USA.
INTRODUCTION

Shallow unconfined groundwater systems are susceptible to contamination from near the ground’s surface, so are not generally used as a source of drinking water in the Salt Lake Valley, Utah, USA (Thiros and Spangler, 2010). In many areas, the shallow aquifer and underlying principal aquifer is separated by less permeable fine-grained sediment which can inhibit the downward movement of water and potential surface contaminants. However, leakage to the deeper aquifer from the shallow aquifer may happen when a downward gradient exists and confining layers are thin and/or discontinuous (Thiros, 2003a). In the Salt Lake Valley, Utah, USA one third of the public water supply is from deep groundwater, while shallow aquifer water is not used for public supply (Thiros, 2003a). Since the deeper aquifer and the shallow aquifer are connected, rendering the deep aquifer susceptible to contamination from the shallow aquifer when a downward gradient exists, shallower groundwater quality needs to be protected to avoid the contamination in deep groundwater (Thiros, 2003a).

Nitrate (NO$_3$-N) contamination to groundwater is a global issue (Hudak, 2000), and has been found throughout the United States (Spalding and Exner, 1993; Nolan et al., 1997; Harter et al., 2002). Drinking water with high NO$_3$-N concentrations can be harmful to human health since high NO$_3$-N concentrations can cause methemoglobinemia in infants and stomach cancer in adults (Addiscott et al., 1991; Wolfe and Patz, 2002). As a result, a maximum contaminant level (MCL) of 10 mg/l NO$_3$-N was established by the U.S. Environmental Protection Agency (USEPA, 2002).
Although NO$_3$-N can occur naturally in groundwater, increased concentrations in groundwater may have resulted from human activities due to increased applications of nitrogenous fertilizers since last century. Nitrogen applied to soils is subject to plant uptake and denitrification. However, when N fertilizer application exceeds plant demand and the denitrification capacity of the soil, N leaching may occur in the form of NO$_3$-N, ultimately reaching groundwater (Almasri and Kaluarachchi, 2004). Agricultural lands receive the most N application, since N is a vital nutrient for enhancing crop production. As a result, agricultural activities are likely the major anthropogenic source of NO$_3$-N contamination to groundwater in agricultural areas (Livingston and Cory, 1998). Similarly, fertilizers applied to urban turfgrass landscapes and gardens may be a source of NO$_3$-N to urban groundwater (Thiros, 2003b), and may pose a hazard to groundwater quality. Ornamental turfgrass landscapes make up a large portion of residential property areas, and soil conditions in the Salt Lake Valley, Utah, USA region often necessitate the application of water and fertilizers to meet turfgrass requirements as well as homeowners’ aesthetic expectations. However, homeowners often over apply such amendments because of a lack of understanding of actual plant needs. Water and fertilizer applied in excess of turf requirements may leach through the soil and contaminate ground and surface waters in communities. Research reviewed by Petrovic (1990) suggested that NO$_3$-N applied to turf areas had the potential to leach through soils and contaminate groundwater if not properly applied. The use of fertilizers on recreational turf landscapes, such as golf courses, has also been identified as a potential source of NO$_3$-N in urban aquifers (Sharma et al., 1996; Wong et al., 1998), as well as turf fertilization in residential areas (Kopp and Guillard, 2005; Saha et al., 2007).
To reduce the N leaching from urban turfgrass landscapes, it is necessary to determine the causal factors of increased groundwater NO$_3$-N concentration. The USGS studied the occurrence and distribution of NO$_3$-N in shallow groundwater underlying areas of recently developed (post 1963) residential and commercial land use in the Salt Lake Valley, Utah, USA based on the assumption that human activities influenced groundwater quality, with results indicating possible human influence on shallow groundwater quality (Thiros, 2003b). Since turfgrass landscapes make up a large portion of residential property areas and may receive excessive amounts of fertilizer, there may be a correlation between groundwater quality and the existence of residential areas around monitoring wells, as has been shown between agricultural land use activities and NO$_3$-N concentration in groundwater of agricultural areas (Keeney, 1989; Wylie et al., 1995; Hudak, 2000; Harter et al., 2002). However, in the Salt Lake Valley, Utah, USA, no correlation was found between the percentage of residential land surrounding the monitoring wells and the concentration of NO$_3$-N in water sampled from the wells in a USGS study (Thiros, 2003b). The absence of correlation between the percentage of residential area and groundwater NO$_3$-N concentration may be due to the fact that turfgrass areas, rather than the entire residential property area, receive the most fertilizer. In addition, the percent of landscaped area on each residential property is different. Soil textures under the landscapes may affect NO$_3$-N leaching as well (Sun et al., 2011). In this study, it was hypothesized that as the percentage of turfgrass area around the monitoring wells increased, the probability of contamination by NO$_3$-N in the well water also increased. Surface soil texture comprised of the largest particle sizes was also hypothesized to increase the probability of NO$_3$-N in the monitoring wells (Burkart et al., 1999; Nolan et al., 2002, Sun et al., 2011). Because no such correlations were found in the USGS
study, a new approach integrating turfgrass area, soil texture and different irrigation and fertilization scenarios was employed to estimate N-leaching from urban landscapes in the Salt Lake Valley.

Various approaches have been used to assess NO$_3$-N leaching to groundwater. For example, assuming a specific fraction of the on-ground N loading will leach as NO$_3$-N (Kim et al., 1993; Cox and Kahle, 1999; Shamruk et al., 2001), conducting simple, efficient N mass balance calculations to estimate the NO$_3$-N leaching to groundwater in agricultural areas (Barry et al., 1993; Goss and Goorahoo, 1995; Puckett et al., 1999), and using soil N models to simulate the N dynamics in the soil (Ramanarayanan et al., 1998). To estimate NO$_3$-N leaching from different soil textures and different management scenarios, a N model is a logical choice. Therefore, a calibrated and verified Hydrus-1D model was utilized to simulate the fate and transport of NO$_3$-N from turfgrass and to determine the mass leaching of NO$_3$-N to groundwater for different soil textures. Spatial analysis techniques are also needed to assess NO$_3$-N leaching from turfgrass areas including different soil textures, and a GIS provides a sound approach to evaluate the NO$_3$-N leaching from various soil textures (Almasri, 2008).

Identification of areas with high N leaching potential is also of importance for land use planners and environmental regulators. When identified, preventive activities can be implemented to decrease the NO$_3$-N leaching risk to groundwater in those identified high-risk areas (Tesoriero and Voss, 1997; Ramanarayanan et al., 1998). Identification of high-risk N leaching areas can pinpoint where groundwater needs to be protected and where improved turfgrass management is most needed.

As a result, the objectives of this research were: (1) to reanalyze the 1999 USGS ground-water NO$_3$-N concentration dataset for NO$_3$-N leaching potential based on a
current Hydrus-1D simulation, (2) to determine whether a relationship exists between potential NO$_3$-N leaching from urban landscapes and ground-water NO$_3$-N concentration, and (3) to find the high NO$_3$-N leaching risk areas in the Salt Lake Valley that may pose potential effects to ground-water quality.

**MATERIALS AND METHODS**

1. **Study Area**

   The Salt Lake Valley is an urban area bounded by the Wasatch Mountain Range, the Oquirrh Mountains, the Traverse Mountains, and the Great Salt Lake. It is 45 km long and 29 km wide and generally corresponds to the populated portion of Salt Lake County, which contains the Salt Lake City metropolitan area. The population of Salt Lake County in 2010 was 1,029,655 (USCB, 2010) and is projected to be 1,223,218 by 2020 (Utah State Data Center, 2000) requiring more water for public supply.

   The climate in Salt Lake Valley is semi-arid with hot summers and moderately cold winters. However, due to the local topography and the large relief between the mountains and valley, the weather can be quite variable (Murphy, 1981). The average annual precipitation is 250-500 mm, mostly in the form of snow. Consequently, lawns and gardens typically require irrigation to supplement natural precipitation during the growing season. The mountains surrounding the valley typically receive substantially more precipitation and have cooler temperatures than the valley, and the southeast part of the county receives the most precipitation (Wallace and Lowe, 2008).
2. Monitoring Shallow Wells

Shallow well NO$_3$-N concentration data from a 1999 USGS study were utilized for this study, and were obtained from the USGS database. The original USGS data were collected in 1999 to quantify relationships between recent residential and commercial areas and groundwater quality (Thiros, 2003b). In the USGS study, “potential well locations were selected by using a computerized, stratified random selection process to ensure that the data collected were unbiased and representative of the quality of water underlying recently developed residential and commercial areas” (Scott, 1990). Forty-one sites in the Salt Lake Valley were selected using the following study criteria:

1. A location in residential and commercial areas developed during 1963-94,
2. A downward gradient between the shallow and deeper aquifers, and,
3. A minimum distance between each site of 1 km.

In the USGS study, more newly developed areas (post 1994) were excluded due to the time necessary for new construction to affect the groundwater quality (Squillace and Price, 1996). Similarly, areas developed before 1963, such as downtown Salt Lake City, were excluded because of the potential for the land use to have changed over time (Thiros, 2003b). The position of the well was determined in latitude and longitude (Figure 1). Shallow ground-water samples were collected in the summer and fall of 1999 (Thiros, 2003b). Nitrate plus nitrite (NO$_2$-N) were detected in samples, and NO$_3$-N was reported as the sum of NO$_3$-N and NO$_2$-N (Thiros, 2003b).
3. Soil Map

The soil map (scale of 1:12000) of the area was obtained from the Soil Survey Geographic (SSURGO) database distributed by the U.S. Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS)-National Geospatial Management Center (NGMC) (Figure 5-1). The SSURGO-certified soils dataset is generally the most detailed level of soil geographic data developed by the National Cooperative Soil Survey. The information was prepared by digitizing maps, by compiling information onto a planimetric correct base and digitizing, or by revising digitized maps using remotely sensed and other information. The data included a detailed, field verified inventory of soils and miscellaneous areas that normally occur in a repeatable pattern on the landscape and that can be cartographically shown at the scale mapped. The soil map was symbolized according to the soil hydraulic conductivities from low to high (Figure 5-1).

4. Growing Season NO$_3$-N Leaching Simulation

A calibrated and validated public domain computer software package (Hydrus-1D) was used to simulate NO$_3$-N leaching from turf grown on different soil textures during the growing season (June to September) under over irrigation and over fertilization scenarios and an efficient irrigation and fertilization scenario (chapter 4). The model simulated soil N transformation and transport in turf using the boundary conditions input, and output including N-leaching from the root zone. All the NO$_3$-N transform and transport parameters were the same as that in the calibration process (chapter 4), and the optimum irrigation and 2010 weather data were used as input boundary conditions to simulate NO$_3$-N leaching under an efficient irrigation and fertilizer management scenario. According to irrigation system evaluation in Salt
Lake City, 150% of efficient irrigation and monthly fertilization at 48.8 kg N/ha rates are typical and were applied in the simulation as over irrigation and over fertilization scenarios. Monthly fertilizer (33-0-0) was applied from June to September at a rate of 48.8 kg N/ha. Fertilizer composition was 2.22% ammonium, 3.93% urea, 8.53% NO$_3$-N, and 18.32% water insoluble nitrogen (WIN). Nitrogen leaching rates for different soil textures were simulated. There are 23 soil textures on the soil map. However, only eight sets van Genuchten parameters for the soil textures were available either in the Hydrus-1D built-in database or from references (Table 5-1). As a result, the NO$_3$-N leaching for these eight soil textures was simulated, and for the rest of the soil textures, N leaching rates were estimated based on the eight simulated soils (Table 5-2). In the simulations, a 15 cm layer of top soil were assumed for all the soil textures based on the assumption that people bring in top soil regardless of existing soil. Furthermore, it was assumed that Kentucky bluegrass was used for landscapes in the valley; NO$_3$-N leached out of root zone (80 cm) will ultimately reach ground water; only turfgrass areas in landscapes received N fertilizer; and turf dominated landscape areas in the valley.

5. Landscape Areas

Green areas in the valley were considered as turf landscape (Figure 5-2). Green pixels were extracted from a satellite image on Aug 3, 1999 to determine the green areas in the map with Normalized Difference Vegetation Index (NDVI) method. The NDVI is a standardized index that allows the generation of an image displaying greenness according to the characteristics of two bands from a multispectral raster dataset—the chlorophyll pigment absorptions in the red band and the high reflectivity of plant materials in the near-infrared (NIR) band.
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NDVI = \frac{(IR - R)}{(IR + R)}

where IR = pixel values from the infrared band, and R = pixel values from the red band. This index outputs values between -1.0 and 1.0, and values between 0.2 to 0.3 representing shrub and grassland, while high values from 0.6 to 0.8 indicate temperate and tropical rainforests. The equation ArcGIS uses to generate the output is as follows:

\text{NDVI} = (\frac{(IR - R)}{(IR + R)}) \times 100 + 100

This results in a value range of 0 to 200 and fits within an 8-bit structure. In this study, 125 < NDVI < 180 were considered as green areas.

6. Predicted NO$_3$-N Leaching Mass

Nitrate leaching from a 500m radius areas around monitored wells was considered to affect well NO$_3$-N concentration since the minimum distance between each site was 1 km. The position of the well in latitude and longitude was determined and all locations were accurate within a 10-m radius. A 500-m radius buffer was constructed around each well location. An ArcGIS script was used to “clip” the soil polygons and the extracted landscape polygons by the 500 m radius buffer. Clipped soil polygons and landscape polygons were intersected and new polygons of soils with landscapes were obtained. Nitrate leaching mass from landscapes around each well was calculated based on soil texture where:

Nitrate-N leaching mass from 500m radius buffer area (kg) = \sum \text{soil areas with landscape (ha)} \times \text{simulated NO$_3$-N leaching rate for each soil type (kg/ha)}

Nitrate-N leaching mass estimation from landscape areas around each monitoring wells was illustrated (Figure 5-3), and the estimated NO$_3$-N mass for 500m radius buffer for each well was shown (Table 5-3).
7. Regression Between NO$_3$-N Concentration in Wells and Estimated NO$_3$-N Leaching Mass Around Each Well

Ground-water NO$_3$-N concentration data were divided into 6 groups according to well depth, the divided groups were 23-36, 38.5, 43.5-48.5, 67.5-77.5, 83.5-92.3, and 95.5-123.5 feet (Table 5-3). Regressions and correlations were developed between ground-water NO$_3$-N concentrations and simulation based NO$_3$-N leaching masses within a 500 m radius around each well. NO$_3$-N concentrations less than 1 mg/L were removed from the regression since those wells were considered not to be affected by human activities. The 153.5 feet deep well was removed from the regression because it was the only well that was much deeper than the 95.5-123.5 feet group.

8. High-Risk Areas

According to the simulated/estimated NO$_3$-N leaching rates from different soils, maps with classes of NO$_3$-N leaching risk areas were developed based on arbitrarily divided NO$_3$-N leaching ranges. Areas with N-leaching of less than 10 kg/ha were considered low risk, 10-25 kg/ha were considered medium risk, 25-40 kg/ha were considered high risk, and higher than 40 kg/ha were considered extremely high-risk areas.

RESULTS AND DISCUSSION

1. NO$_3$-N Concentration of Shallow Residential Well Water

It has been reported that background NO$_3$-N concentrations in groundwater from areas not associated with agricultural management practices are commonly less than 2 to 3 mg L$^{-1}$ (Hallberg and Keeney, 1993). As such, NO$_3$-N concentrations greater than 2 mg L$^{-1}$ may indicate groundwater quality affected by human activities (USGS, 1999). The USGS shallow groundwater NO$_3$-N concentration data showed that 86.7%
(26 of 30) of monitoring wells had NO$_3$-N concentrations higher than the assumed background level of 2 mg L$^{-1}$, suggesting a possible human influence on shallow groundwater quality (Table 5-3). The high frequency of monitoring well NO$_3$-N concentration exceeding background levels in the residential areas may have resulted from the application of nitrogenous fertilizers that ultimately leached as NO$_3$-N (Thiros, 2003b). The median NO$_3$-N concentration of the 30 samples was 6.85 mg L$^{-1}$, and the concentrations ranged from less than 0.05 to 13.3 mg L$^{-1}$ (Table 3). Ten percent (3 of 30) of the monitoring wells exceeded the USEPA MCL of 10 mg L$^{-1}$ NO$_3$-N in drinking water (USEPA, 2002) (Table 5-3).

2. Correlation Between NO$_3$-N Concentration in Wells and Estimated NO$_3$-N Leaching Mass Around Each Well

Although landscape areas and soil textures were included in this approach to estimate NO$_3$-N leaching, no correlation between ground-water NO$_3$-N concentration and estimated NO$_3$-N leaching mass was found when all well ground-water NO$_3$-N concentration data were included, and this finding supports the conclusion of the 1999 USGS study that there was no relationship between the percentage of residential land use surrounding the monitoring wells and the concentration of NO$_3$-N in water sampled from the wells (Thiros, 2003a). This lack of correlation may have resulted from the ground-water flow that mixed the ground water. Another factor may be the limited size of the dataset of ground-water NO$_3$-N concentrations, and the small range of NO$_3$-N concentrations.

Well depth may also play a role in the lack of correlation since it takes time for NO$_3$-N to reach deep ground water, allowing more time for NO$_3$-N to be subjected to denitrification or other loss processes. Therefore, NO$_3$-N concentrations were grouped by well depth, and correlations between ground-water NO$_3$-N concentration and
simulated NO$_3$-N leaching mass were developed according to well depth groups (Table 5-2, Figure 5-4). Although only 3-6 wells were assigned to each group, correlations were found between soil NO$_3$-N concentration and simulated NO$_3$-N leaching mass for the 38.5, 67.5-77.5, 83.5-92.5, and 95.5-123.5 feet groups, with $R^2$ values of 0.42, 0.65, 0.47, and 0.47 for each group, respectively. This suggests that landscape areas and soil textures had some influence on NO$_3$-N leaching from the root zone, and finally affected the ground-water quality to a certain extent, supporting the hypothesis that coarse soil textures may result in increased NO$_3$-N leaching to ground water under landscape areas. The stronger correlations of the 67.5-77.5 ft depth and 95.5-123.5 ft depth resulted from a few points with high simulated NO$_3$-N leaching mass, and this high leaching mass estimation resulted from certain areas with extremely high nitrate leaching rates, for example, very cobbly loamy sand. A small percentage of high leaching rate areas can result in high leaching mass for the 500 m radius areas around each well of this study, and this result suggests that soil texture is the determining factor in NO$_3$-N leaching estimation although estimated NO$_3$-N leaching mass depended on both soil textures and landscape areas.

In addition to the limited size of the ground-water NO$_3$-N concentration dataset and the potential mixing process in ground water, there are some other potential reasons for the lack of a strong correlation between ground-water NO$_3$-N concentrations and estimated NO$_3$-N leaching. First, the simulation was based on assumptions that all the landscape areas were over irrigated and over fertilized, which is very common, but not true for each and every landscape. Second, urban fertilization may not be the only NO$_3$-N source to ground water in urban areas. Septic leakage, sewer leakage, or landfill leakage may also play a role in NO$_3$-N contamination (Thiros, 2003a; Wakida and Lerner, 2005). Furthermore, contamination may even
result from well construction or other factors connected to ground-water quality that was not investigated in the current study.

3. Ground-water NO$_3$-N Concentration and Well Depth

In addition to the shallow ground-water NO$_3$-N data from USGS (1999), NO$_3$-N concentration data from another 30 deep wells were considered in Figure 5-5 (Wallace and Lowe, 2008). It may be expected that shallow wells are more susceptible to contamination than deeper wells, and this was confirmed by the plot of shallow and deep well NO$_3$-N concentration vs. well depth (Figure 5-5). In shallow ground water (depth <50 m) NO$_3$-N concentration ranged from 0.2 to 13.3 mg/l. However, in deep wells (>50 m), none of the well NO$_3$-N concentrations exceeded the USEPA MCL limit of 10 mg/l and most of the well NO$_3$-N concentrations were less than 4 mg/l. This finding indicates that while NO$_3$-N was able to contaminate deep groundwater, shallow groundwater was more susceptible to NO$_3$-N contamination. When NO$_3$-N concentrations in deep groundwater are elevated, it may be due to leakage from the shallow aquifer to the deeper principal aquifer, since leakage is possible where a downward gradient exists and confining layers are thin and/or discontinuous (Thiros, 2003a).

In the Salt Lake Valley, water from the deeper aquifer underlying the shallow ground-water system is used for the public drinking water supply (Thiros, 2003a). The NO$_3$-N concentrations less than 10 mg L$^{-1}$ in deep wells indicates that deep groundwater in the Salt Lake Valley is safe for drinking, when NO$_3$-N concentration is the concern. The low NO$_3$-N concentrations in deep wells may be affected by several factors. For example, the amount of time required for NO$_3$-N to reach deep groundwater results in a greater opportunity for denitrification. Additionally, leaked NO$_3$-N from shallow groundwater is diluted in the larger volumes of deep
groundwater. And while the shallow aquifer is susceptible to surface contamination from land use activities because of its proximity to the land surface, the deeper unconfined aquifer is vulnerable because of a lack of confining layers that can impede the downward movement of contaminated groundwater (Thiros, 2003a).

4. Risk Area

Class of risk area maps were developed for urban areas in the Salt Lake Valley under efficient irrigation and fertilization management scenarios and over irrigation and over fertilization scenarios. Under conditions of over irrigation and fertilizer, 20% of urban areas have high (25-40 kg/ha NO$_3$-N leaching) or extremely high risk (>40 kg/ha NO$_3$-N leaching) of contamination by NO$_3$-N leaching from urban landscapes, while 48% and 17% of urban areas have medium or low contamination risk, respectively (Figure 5-6). However, under efficient management, most of the urban areas are at low risk of contamination, meaning less than 10 kg/ha NO$_3$-N can be leached out of root zone (Figure 5-7). Under these conditions, 83% of these areas have low contamination risk, while only 1% have medium contamination risk. No high risk or extremely high risk areas exist under efficient management scenarios.

Studies have illustrated that groundwater is closely connected to the landscape and land use that it underlies, and is vulnerable to the management of the land surface above (Harter et al., 2002; Lerner and Harris, 2009). Recharge to groundwater and the use of groundwater can affect groundwater quality and quantity, and are determined by land use and management. As a result, inappropriate land use and poor land management may cause chronic groundwater quality problems (Lerner and Harris, 2009).

However, even if efficient management strategies are implemented in urban landscapes, immediate decreases in NO$_3$-N leaching to groundwater may not be
possible because of the pool of N existing in soil (Almasri and Kaluarachchi, 2004). Research has shown that NO\textsubscript{3}-N leaching continued even after the termination of operations and reduction in N loading in livestock feedlots, for example (Gormly and Spalding, 1979; Carey, 2002). And even when NO\textsubscript{3}-N leaching from agricultural areas to groundwater decreases or stops immediately due to improved practices, groundwater NO\textsubscript{3}-N concentrations will not drop immediately (Lerner and Harris, 2009).

Some studies have found persistent groundwater N concentrations after NO\textsubscript{3}-N contamination was stopped and management alternatives were in place for as long as 30 years (Gelhar and Wilson, 1974; Mercado, 1976, Hudak 2000; Shamrukh et al., 2001; Nolan et al., 2002; Wakida and Lerner, 2002), confirming that groundwater NO\textsubscript{3}-N concentrations do not drop immediately as a result.

5. Considerations of the Study

The interactions of land use, on-ground N loading, irrigation management, recharge, N dynamics, soil characteristics, and depth of soil are complex, so it is difficult to quantify NO\textsubscript{3}-N leaching accurately (Almasri, 2007). Given this complexity and difficulty, the results of this study must be carefully evaluated and considered prior to making consequential policy or management decisions based on the findings.

One consideration results from the NO\textsubscript{3}-N transport and transformation parameters. It has been demonstrated that soil type can affect N transformation rates (Shearman, 1986), and that soil transformation processes (mineralization/immobilization, nitrification, denitrification, and plant uptake) greatly affect NO\textsubscript{3}-N leaching. Soil characteristics dictate N kinetics as well. For example, in well-drained soils with high infiltration, the rate of nitrification is high and
denitrification may be insignificant. In contrast, in poorly drained soils, denitrification is high and nitrification may be insignificant (Almasri, 2007). In this study, nitrification and denitrification parameters were held constant for all the soil texture scenario simulations to estimate NO$_3$-N leaching from different soils. Furthermore, soil depth controls the time lag between on-ground applications of N and NO$_3$-N leaching, and influences the time span of soil N transformations (Almasri and Kaluarachchi, 2004). As a result, the NO$_3$-N leaching mass estimation for different soil textures is subject to some uncertainty.

Another consideration results from the soil textures of the soil survey map. The soil survey map was based on the top 2 m soil texture, and soil textures deeper than 2 m were unknown. Although in this study the NO$_3$-N leaching estimation was based on simulated NO$_3$-N leaching from the top 80 cm soil, the unknown soil textures deeper than 2 m may decrease NO$_3$-N leaching, or may even stop NO$_3$-N leaching if a confining layer exists.

Other considerations relate to the assumptions made in the study. For example, it was assumed that all property owners/managers bring in 15cm of top soil. It was further assumed that NO$_3$-N leaching beyond the turfgrass root zone would reach groundwater. However, NO$_3$-N leaching out of root zones is subject to denitrification and denitrification rates depend on soil texture and soil depth when temperature and moisture content are the same. In addition, all the landscape areas were assumed to be covered with turf. However, trees and shrubs are also common in landscapes and NO$_3$-N leaching out of turf root zones may be absorbed by shrubs and trees which have much deeper root systems and may decrease NO$_3$-N leaching to groundwater.
CONCLUSIONS

Although there were many assumptions made in this study, the proposed methodology of integrating soil textures and N modeling was useful for estimating NO\textsubscript{3}-N leaching from urban landscapes in the Salt Lake Valley, and it was validated with measured ground-water NO\textsubscript{3}-N concentrations to some extent. Deep ground water had much lower NO\textsubscript{3}-N concentrations than shallow ground water, and shallow ground water is more susceptible to surface contamination. However, shallow ground water contaminants are able to reach deep ground water and decrease the deep ground-water quality under conditions of lacking confining layers. The results of this study indicate that improvement of turf irrigation and fertilization management may decrease N-leaching significantly and greatly decrease the risk of ground water being contaminated by NO\textsubscript{3}-N leaching in the Salt Lake Valley, although such management changes cannot immediately halt or reverse the consequences of past NO\textsubscript{3}-N leaching.

LITERATURE CITED


Leaching from Fertilizer Applied on Golf Course: Lysimeter Study. Water Air
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TABLE 5-1. van Genuchten parameters for different soil textures used in the Hydrus-1D simulation.

<table>
<thead>
<tr>
<th>Soil textures</th>
<th>( \theta_r )</th>
<th>( \theta_s )</th>
<th>( \alpha ) (1/cm)</th>
<th>( n )</th>
<th>( K_s ) (cm/d)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coarse sandy loam</td>
<td>0.057</td>
<td>0.41</td>
<td>0.124</td>
<td>2.28</td>
<td>350.2</td>
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<tr>
<td>Loam</td>
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<td>0.43</td>
<td>0.036</td>
<td>1.56</td>
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<tr>
<td>Fine sandy loam</td>
<td>0.112</td>
<td>0.44</td>
<td>0.009</td>
<td>2.873</td>
<td>100.8</td>
</tr>
<tr>
<td>Sand</td>
<td>0.045</td>
<td>0.43</td>
<td>0.145</td>
<td>2.68</td>
<td>712.8</td>
</tr>
<tr>
<td>Gravelly loam</td>
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<td>0.47</td>
<td>0.09</td>
<td>1.46</td>
<td>50</td>
</tr>
<tr>
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</tr>
<tr>
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<td>0.01</td>
<td>1.23</td>
<td>1.68</td>
</tr>
<tr>
<td>Sandy loam</td>
<td>0.065</td>
<td>0.41</td>
<td>0.075</td>
<td>1.39</td>
<td>106.1</td>
</tr>
</tbody>
</table>
TABLE 5-2. Simulated/estimated N-leaching rates for Kentucky bluegrass under efficient irrigation and fertilization (100%), and inefficient irrigation (150%) and fertilization (200%) scenarios for soils of the survey map.

<table>
<thead>
<tr>
<th>Soil Texture</th>
<th>Efficient irrigation and fertilization</th>
<th>Inefficient irrigation and fertilization</th>
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</thead>
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<tr>
<td></td>
<td>Simulation</td>
<td>Estimation</td>
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<td>Coarse sandy loam</td>
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<td>Loam</td>
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</tr>
<tr>
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<td>14</td>
</tr>
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<td>Sand</td>
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</tr>
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<td></td>
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<td>19.5</td>
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<td>10</td>
</tr>
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<td></td>
<td>Sand loam</td>
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</tr>
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<tr>
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<tr>
<td></td>
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</tr>
<tr>
<td></td>
<td>9</td>
<td>55</td>
</tr>
<tr>
<td></td>
<td>Very cobbly loam</td>
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<td>50</td>
</tr>
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<td>35</td>
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<tr>
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<td>Very cobbly loam sand</td>
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<tr>
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</tr>
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</tr>
<tr>
<td></td>
<td>Very gravelly sandy</td>
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</tr>
<tr>
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TABLE 5-3. Grouping of wells, well depth, NO$_3$-N concentration, landscape areas in 500m radius around wells, and estimated N leaching mass from 500 m radius landscape areas of each well in 1999.

<table>
<thead>
<tr>
<th>Groups</th>
<th>Well Depth</th>
<th>NO$_3$-N concentration (mg/l)</th>
<th>Landscape areas around wells (ha)</th>
<th>Sum NO$_3$-N leaching from landscape areas (kg)</th>
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<td>4.14</td>
<td>42.5</td>
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<td>38.5</td>
<td>12.7</td>
<td>30.1</td>
<td>366</td>
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<td>38.5</td>
<td>3.55</td>
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<td>5.46</td>
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<td>2.37</td>
<td>23.5</td>
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<td>7.35</td>
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<td>4.72</td>
<td>48.8</td>
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<td>43.5</td>
<td>7.05</td>
<td>15.5</td>
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<td>48.5</td>
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<td>304</td>
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<td>31.5</td>
<td>0.2</td>
<td>46.1</td>
<td>1230</td>
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<td>1740</td>
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<td>77.5</td>
<td>0.25</td>
<td>46.1</td>
<td>1230</td>
</tr>
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</table>
FIGURE 5-1. Location of shallow monitoring wells and soil maps in the urban areas of Salt Lake valley, Salt Lake County, UT.
FIGURE 5-2. Location and NO$_3$-N concentration of shallow monitoring wells and landscape areas (1999) in urban areas of the Salt Lake Valley (Salt Lake County, UT).
FIGURE 5-3. Landscape area within 500 m radius of each monitoring well with soil textures, and the estimation of NO$_3$-N leaching mass from the landscape areas.
FIGURE 5-4. Relationship between depth grouped monitoring well ground-water NO\textsubscript{3}-N concentrations and simulation based N-leaching masses/500m radius buffer around monitoring wells (Salt Lake Valley, Salt Lake County, UT).
FIGURE 5-5. NO$_3$-N concentration of both deep and shallow wells in the Salt Lake Valley (Salt Lake County, UT). Shallow well data were from 1999, and deep well data were from 2001.
FIGURE 5-6. Risk class of urban ground water being contaminated by NO$_3$-N leaching from urban landscapes according to soil textures above ground water under overirrigation and overfertilization scenarios in the Salt Lake Valley (Salt Lake County, UT).
FIGURE 5-7. Risk class of urban ground water being contaminated by NO$_3$-N leaching from urban landscapes according to soil textures above ground water under efficient irrigation and fertilization management scenarios in the Salt Lake Valley (Salt Lake County, UT).
CHAPTER 6
CONCLUSIONS

Under the well-watered conditions of a drainage lysimeter study, this research determined that plant canopy cover—rather than plant material water use categorization—was the controlling factor in woody plant and perennial water use. Consequently, homeowners and other landscape managers may achieve meaningful water savings by simply adjusting planting densities in the landscape. In the meantime, adjusting the landscape areas of woody plants, turf and perennials based on $K_p$ and adjusted $K_p$ of each hydrozone may provide another method for conserving water in landscapes. The $K_p$ values and the irrigation timings for different plant types developed from this study may also serve as a guideline for setting well-watered irrigation schedules in the IMW region. Under water-stressed conditions, however, plant material choice will likely play a more central role in overall landscape water use. The $K_z$ values developed, based on the relationship between $K_p$, canopy cover and drought tolerance factors for woody plants and perennials in the Mesic, Mixed, and Xeric landscapes in this study, may serve as guidelines for setting irrigation schedules in the IMW region for water efficient landscapes. Further research on landscapes under water-stressed conditions is needed to ascertain the drought tolerance factors for different plant types under different drought tolerance categorizations.

In the simulation study, calibration and validation results showed that Hydrus-1D may be used for simulation of water and N distribution and leaching in landscapes. Scenario simulation results revealed that over irrigation is the main contributing factor to NO$_3$-N leaching, and that NO$_3$-N leaching increased as irrigation increased. Over
fertilization alone, with optimized irrigation, did not increase NO$_3$-N leaching during the growing season. However, under conditions of over irrigation and over fertilization, NO$_3$-N leaching increased significantly. Soil textures had a significant effect on N leaching as well. In a coarse-textured, sandy loam soil, more N may be leached than loam and clay loam soils under the same irrigation and fertilization conditions. Therefore, from the scenario simulation results, homeowners and turfgrass managers should pay the most attention to irrigation schedules coupled with fertilizer application, especially on sandy loam (and sandy) soils, to decrease the potential for NO$_3$-N leaching.

The GIS method of integrating soil texture with a N model utilized in this study is capable of estimating NO$_3$-N leaching from urban landscapes in the Salt Lake Valley, and it was validated to some degree with measured ground-water NO$_3$-N concentration. Deep ground water had much lower NO$_3$-N concentrations compared to that of shallow ground water, and shallow ground water was more susceptible to surface contamination. However, shallow ground water contaminants are able to reach deep ground water and decrease deep ground-water quality under conditions of lacking confining layers. Improvement of turf irrigation and fertilization management can decrease N-leaching significantly and greatly decrease the risk of ground water being contaminated by NO$_3$-N leaching in the Salt Lake Valley, although such improved management cannot stop NO$_3$-N leaching to ground water immediately.
Curriculum Vitae

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(October 2011)

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Research Interests

- Water efficient landscaping, Plant drought tolerance, Plant cultivation
- Nitrate leaching, Nitrogen transport and transform simulation
- Ground water quality

Education

Doctor of Philosophy
Plant Science, Utah State University
Area: Water efficient landscaping
Dissertation: Characterizing Water and Nitrogen Dynamics in Urban/Suburban Landscapes
Advisor: Dr. Kelly Kopp

Master of Agronomy
Ornamental Horticulture, China Agricultural University
Area: Plant Nutrition
Thesis: the Influence of N-form (NH₄⁺ vs NO₃⁻) and Application Amount on the Growth of Limonium bicolor
2004 - 2007

Bachelor of Agronomy
Landscape Architecture, Henan Agricultural University
1999-2003

Bachelor of Economics
Agricultural Economics, Henan Agricultural University
1999-2003

Publications and Presentations

Publications


Presentations


TRAINING EXPERIENCE

Hydrus Short Course. 2010. Colorado School of Mines, Golden, CO

RESEARCH SKILLS

Good command of experimental techniques related to TDR, TDT soil moisture sensors, CR1000 Datalogger, flowmeters, leaching collection.
Good command of GIS, Hydrus-1D, CR Basic programming.
Strong ability of analyzing data with EXCEL, SAS, SigmaPlot, Table Curve.