Soil Biogeochemical Consequences of the Replacement of Residential Grasslands with Water-Efficient Landscapes

by

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ABSTRACT

As a result of growing populations and uncertain resource availability, urban areas are facing pressure from federal and state agencies, as well as residents, to promote conservation programs that provide services for people and mitigate environmental harm. Current strategies in US cities aim to reduce the impact of municipal and household resource use, including programs to promote water conservation. One common conservation program incentivizes the replacement of water-intensive turfgrass lawns with landscapes that use less water consisting of interspersed drought-tolerant shrubs and trees with rock or mulch groundcover (e.g. xeriscapes, rain gardens, water-wise landscapes). A handful of previous studies in experimental landscapes have shown that converting a turfgrass yard to a shrub-dominated landscape has the potential to increase rates of nitrate (NO_3^-) leaching. However, no studies have examined the drivers or patterns across diverse management practices. In this research, I compared soil nutrient retention and cycling in turfgrass and lawn-alternative xeriscaped yards along a chronosequence of time since land cover change in Tempe, Arizona, in the semi-arid US Southwest. Soil inorganic extractable nitrogen (N) pools were greater in xeriscapes compared to turfgrass lawns. On average xeriscapes contained 2.5±0.4 g NO_3^--N/m^2 in the first 45 cm of soil, compared to 0.6±0.7 g NO_3^--N/m^2 in lawns. Soil NO_3^--N pools in xeriscaped yards also varied significantly with time: pools were largest 9-13 years after cover change and declined to levels comparable to turfgrass at 18-21 years. Variation in soil extractable NO_3^--N with landscape age was strongly influenced by management practices that control soil water availability, including shrub cover, the presence of subsurface plastic sheeting, and the frequency of irrigation. This research is the first to explore the ecological outcomes and temporal dynamics of an increasingly common, ‘sustainable’ land use practice that is universally promoted in US cities. Our findings show that transitioning from turfgrass to water-efficient residential landscaping can lead to an accumulation of NO_3^--N that may be lost from the soil rooting zone over time, through leaching following irrigation or rainfall. These results have implications for best management practices to optimize the benefits of water-conserving residential yards.

TABLE OF CONTENTS

CHAPTER

CHAPTER Page

LIST OF TABLES

Table Page

LIST OF FIGURES

Figure Page

1. Introduction

In the United States (US), the primary location of direct human-environment interactions is the urban landscape. Urbanized areas, covering approximately 3-5% of the land, are home to over 70% of people in the US (Census Bureau 2010). The concentrated populations in urban and suburban areas and uncertainty in local and regional resource availability have lead cities and municipalities to adopt conservation programs that aim to both provide services for people and mitigate environmental harm (Grimm et al. 2000, Opp et al. 2013). Current strategies in many US cities aim to reduce the impact of municipal and household resource use such as essential programs to promote water conservation.

Water conservation is a critical sustainability goal in urban areas worldwide, and is particularly relevant for cities in arid and semi-arid climates such as the drought-prone western and southwestern (SW) US (NDRC 2008, Hilaire 2009, NOAA 2014). Weather extremes, increasing population density, and the additive effects of urban heat islands create challenges in urban water management for even the most water-secure cities (Morehouse 2000, Vörösmarty et al. 2010). Since the late 1960's, water conservation programs have been developed to reduce residential water use (Gleick 2014). Because 30-50% of household in the US is consumed outdoors for landscaping (EPA 2015) conservation programs that target outdoor irrigation are key to reducing urban water consumption (Balling et al. 2008).

One increasingly common conservation program in cities incentivizes the replacement of water-intensive turfgrass lawns with more water-efficient landscapes that typically consist of interspersed drought-tolerant shrubs and trees with rock or mulch groundcover (e.g. in different regions called xeriscapes, rain gardens, ornamental gardens, climate appropriate landscapes, water-wise landscapes, and others). In 2015, 77 cities in 16 US states offered financial incentives to encourage homeowners to remove and/or replace their lawns. For example, once known as the 'green oasis in the desert', the Phoenix (Arizona) metropolitan area, was dominated by turfgrass lawns and broad-leaf trees in the 1960's, but in 2005 approximately 66% of yards were landscaped in drought-tolerant vegetation (Stefanov et al. 2001, Gleick et al. 2003, Buyantuyev 2010, Denver Water 2013). This radical land cover change from turfgrass lawns to mixed shrubs and trees reduces water use as intended and often fertilizer use as well (Sovocool et al. 2006, Hilaire 2009) but may also have unintended consequences for water quality (Amador et al. 2007). Studies in native arid and semi-arid ecosystems show that land cover change from grassland to patchy shrubland leads to loss of nitrogen (N) from the plant-soil system through erosion, runoff, and leaching. These losses are due to reduction in soil stabilization by roots, increased rates of surface water runoff relative to infiltration, changes in soil moisture, and inconsistent plant nutrient uptake (Parsons et al. 1996, Baer et al. 2006, Turnbull et al. 2010). Construction of shrubland landscapes from fertile, managed urban grasslands may lead to similar biogeochemical outcomes, which could compromise water resource sustainability objectives.

While US cities expand in population and area, turfgrass cover continues to be the land cover of choice for homeowners and businesses outside of the SW (Zhang et al. 2015). Urban grasslands cover more land area than all the major US crops combined, including barley, cotton, and even corn (Milesi et al. 2005, Zhou et al. 2008). Grassy lawns provide numerous aesthetic benefits (Beard et al. 1994), and can also mitigate the urban heat island effect through evaporative cooling (Jenerette et al. 2011), and sequester high amounts C due to relatively high net primary productivity from to supplemental water and fertilizer (Pouyat et al. 2006). However, as the most irrigated land cover type in US, grass lawns require substantial water inputs (on average 1 L/m^2 per day) (USDA NIFA 2011), and they consume 7-10 times more water in arid and semi-arid climates than in more mesic regions (USDA NASS 2003, Milesi et al. 2005). Furthermore, turfgrass yards are typically managed intensively by homeowners, who spend time and money to irrigate and apply insecticides, herbicides, and fertilizer as well as use equipment that burns fuel (Robbins & Birkenholtz 2003). Numerous studies have found that lawn-alternative, shrub-dominated landscapes generally require 35-75% less water than turfgrass yards (McPherson et al. 1990, Sovocool et al. 2006). However, barriers to technology adoption and use (e.g. installation of drip irrigation and appropriate use of irrigation timers) and other drivers of human behavior continue to challenge the notion that climate-appropriate landscapes always lead to significant water savings (Erickson et al. 2001, Wentz 2007, Martin 2008).

As urban land cover continues to expand, the ubiquity of intensively managed residential landscapes has led to concerns about water pollution due to surface runoff and leaching of soil nutrients (Petrovic 1990, Morton et al. 1988, Lehmann & Schroth 2003). Urban grasslands can contain as much N as agricultural soils due to intensive nutrient inputs, held mostly within a dense network of actively growing roots (Baer et al. 2002, Zhu et al. 2006, Raciti et al. 2008). N compounds such as NO_3^--N are highly mobile in the soil column and contribute to contamination and eutrophication of both ground water and aquatic ecosystems in highly managed agricultural and urban areas Paul & Clark 1989, Matson et al. 1997). Some studies have shown that N losses can be high from lawns, depending on rates of fertilization and landscape age (Gold et al. 1990, Engelsjord 1997, King et al. 2001, Guillard & Kopp 2004, Easton & Pretovic 2004, Shi et al. 2006, Raciti et al. 2011), Turfgrass nutrients are particularly vulnerable to loss during establishment but are more likely to be held in the soil over time due to accumulation of soil organic matter (McClellan et al. 2009). However, recent research shows that fast-growing turfgrass lawns actually retain more nutrients than previously thought, maintaining small pools of mobile N in soils and supporting surprisingly low rates of NO_3^--N leaching (Martin 2001, Zhu et al. 2006, Groffman et al. 2009, Martinez et al. 2014).

Much less is known about the fate of soil nutrients in water-conserving landscapes, particularly relative to the turfgrass lawns these landscapes replace. One study in experimental plots concluded that shrubs are more effective at using water and nutrients than turfgrass (Qin et al. 2013). However, a handful of other studies show that experimental alternative landscapes (gardens with wood mulch and shrubs) have the potential to lose more N than grass landscapes, both during the grass-to-shrub transition and when plants are mature (Amador et al. 2007, Loper et al. 2013). For example, mixed-species ornamental landscapes supported 10-fold higher leaching rates than turfgrass soon after establishment, comparable to rates under corn (*Zea mays L.*; Loper et al. 2013; 48.3 kg N/ha ornamental vs 4.1 kg N/ha turfgrass, Erickson et al. 2001). After turfgrass death, rates of NO_3^--N leaching are high due to diminished plant uptake and changes in microbial activity (Jiang et al. 2000, Hull et al. 2001). In arid and semi-arid climates, soil nutrients accumulate during long dry periods in shrubland ecosystems and are subject to rapid transformation and loss after pulsed precipitation events (Walvoord et al. 2003, Austin et al. 2004, Reichmann et al. 2013). In a recent study, Hale et al. (2014), found that watershed-scale N export in storm water was higher from neighborhoods composed of desert-style landscaping compared to neighborhood with a high proportion of turfgrass yards.

Despite the growing prevalence of climate-appropriate landscapes, no studies to date have characterized the ecological outcomes of this common land cover change in heterogeneous residential urban or suburban areas. In this study, I explore soil properties and nutrient cycling across an urban land cover change from grassland to shrubland. I hypothesized that the replacement of turfgrass with a climate-appropriate landscape (hereafter referred to as ‘xeriscapes’) would create disturbed moist soils that would favor mineralization of organic N, nitrification and mobilization of NO_3^--N due to limited and heterogeneous N uptake by shrubs. Furthermore, I hypothesized that soil nutrient content would decrease with xeriscape age (time since land cover change) as water inputs cause movement of nutrients down the soil column, or as nutrients are taken up by maturing vegetation. I tested these hypotheses across a chronosequence of time since turfgrass removal in yards of single-family homes in the City of Tempe, AZ, located in metropolitan Phoenix. Because homeowners and hired landscapers determine the structure, vegetative composition, and maintenance of residential landscapes, I hypothesized that N-cycling in these extraordinarily heterogeneous landscapes would function similarly when homeowner management was similar across yards, such as frequency of irrigation, cover of vegetation (%), and procedures used during the land cover change process (Paul & Clark 1989, Austin et a. 2004, Loper et al. 2013). Increased understanding of the patterns and drivers of yard nutrient dynamics will help shape best management practices to achieve multiple sustainability outcomes in urban and suburban areas.

2. Experimental Design and Methods

2.1 Survey Method, Site Selection, and Sample Design

I explored the soil biogeochemical outcomes of a residential land cover change from turfgrass to xeriscape across a chronosequence xeric landscape age in the City of Tempe, Arizona (USA). Tempe is a city of 168,000 residents located in metropolitan Phoenix within the Sonoran Desert (population 4.5 million, Census Bureau 2014, US Census Bureau 2010). The climate is arid, with annual precipitation at 18.3 cm split between two rainy seasons (Maricopa Country, AZ 2015). The summer monsoon season, typically occurring from July to September, is characterized by intense winds and sporadic and short heavy rains that can make up 50% of the annual rainfall. The winter rainy season brings longer, lighter rainfall events (Guido 2008). Study area soils are Avondale or Laveen clay loam derived from mixed alluvium parent material with a slope from 0-1 percent (USDA NRCS 2013). All residential sites used in this study have an agricultural land legacy of diversified crops including cotton, corn, and citrus (Knowles-Yanez et al. 1999). The temperatures range from average highs of 30° C to average lows of 12° C (NOAA 2015).

The City of Tempe is also located within the boundaries of the 6400 km^2 Central Arizona Long Term Ecological Research site (CAP-LTER). In 1993, Tempe was one of the first cities in the nation to offer financial rebates to replace turfgrass lawns with drought-tolerant shrubs. Since then, the city has kept records of participants in the Tempe Landscape Rebate Program, documenting the address, homeowner name, as well as yard area and rebate year. Common yard plants in xeriscapes include both native and non-native species of cacti, such as saguaro (*Carnegiea gigantea*), prickly pear (Genus *Opuntia*), and barrel cacti (Genus *Echinocactus*); shrubs, such as creosote (*Larrea tridentata*), brittlebush (*Encelia farinose*), and Mexican bird of paradise (*Caesalpinia pulcherrima*), among others, and N-fixing trees, such as mesquite (*Prosopis chilenses & Prosopis velutina*) and acacia (*Acacia berlandieri* & *Acacia constricta*) and palo verde trees (*Parkinsonia florida* & *Parkinsonia microphylla)* (USDA NRCS 2015).

2.2 *Sampling Design*

In order to explore the fate and temporal dynamics of soil N after turfgrass removal, I sampled soil and yard properties from 47 single-family homes in Tempe that were landscaped in their front yards with either turfgrass (time since land cover change=0) or desert-style xeriscapes (time since conversion=1-21 yrs). I selected 5 houses with turfgrass lawns using convenience sampling. I then selected 1400 homes that participated in the Tempe Landscape Rebate Program, stratified by the year they received the rebate so that an equal number of homes were selected at random from each year. In April of 2014, I sent postcards to the homes with a link to an online survey to gather data on yard history and management, and secure homeowner permissions for in-situ research.

The response rate of the household surveys was 10% (n=140) and of those, 40 individuals opted in for further study and their yards were used for soil sampling. All sites are single-family homes in the City of Tempe that have an agricultural land legacy. In order to avoid location bias I mapped the addresses using GIS software and visually assessed that the sites were spatially distributed across the city. I further grouped the participating households (n=47) into age categories containing an approximately equal number of yards: <4, 4-8, 8-13, & 18-21 years since land cover change, & turfgrass yards (0 years since land cover change). This design provided multiple replicates for each category including grassy lawns as age zero (n=5-8 per category) (Figure 1). Xeric yards in the study varied greatly in their size, structure, and vegetative composition (Table 1) as well as management.

I included survey questions that fell under three general categories: current yard cover and maintenance, the yard conversion process and motivations, and demographics (See appendices). These categories provided information on how the homeowner managed his/her yard, as well as common practices used during the conversion process. I contacted individuals by post-card, written in English, which contained a link to the online survey. Homeowners were also given the option to participate in the survey over the phone or by filling out a hard copy sent to their home.

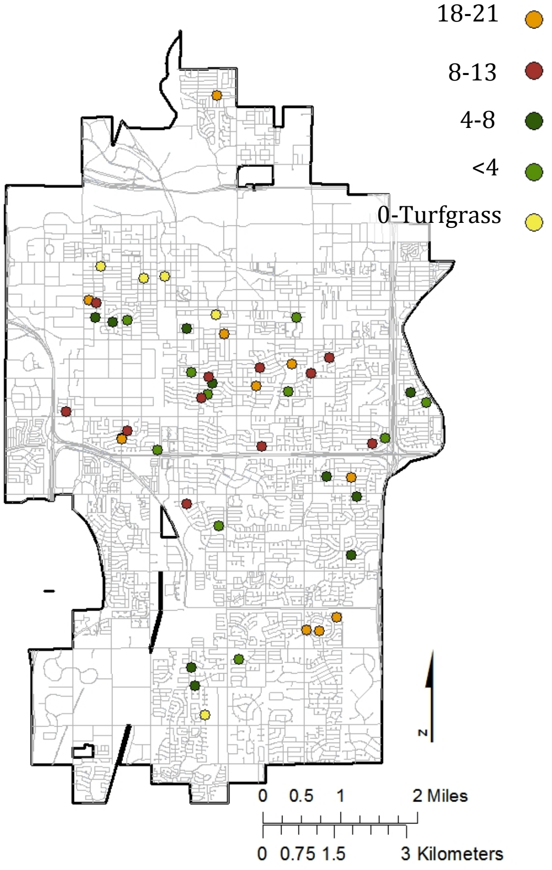

Figure 1. Map of Tempe, AZ with study sites indicated by year since land cover change to xeriscape and turfgrass yards (n=5-8 houses per category). All sites fall in areas that were previously in agriculture and have similar soil types.

Years since land cover change	**Cover of shrubs**	**Canopy cover of trees**	**Canopy cover of N-fixing trees**	**Site size**
Years	%	%	%	m^2
Turfgrass	11.1 (4.68)	60.24 (8.29)	18 (13.35)	290.25 (67.59)
<4	16.81 (4.49)	43.71 (6.66)	26.02 (9.15)	196.87 (15.05)
4-8	17.98 (3.19)	57.03 (17.68)	25.05 (9.33)	255.97 (33.22)
8-13	7.25 (2.47)	56.9 (10.87)	27.78 (11.66)	205.16 (22.33)
18-21	15.09 (2.76)	57.92 (9.57)	42.1 (9.96)	185.69 (11.56)

Table 1. Site characteristics, divided by categories of years since land cover change. Values represent mean (± SE).

2.2 *Yard Vegetation*

Drought-tolerant shrubs, cacti, and desert-adapted trees are common in xeriscapes in Tempe, but the number of plants and level of maturity is highly variable among the study sites (see Appendix B).

In order to identify potential effects of plant nutrient uptake on nutrient pools and fluxes I quantified the area of ground cover, and canopies of shrubs and trees in the front yards of each home using visual surveys. For ground cover, I segmented yards into four quadrants and recorded the approximate cover of grass, gravel, bare soil, or impervious (pavement/stones), summing to 100% for each quadrant. The data for each quadrant were later multiplied by 0.25 and summed together to get a whole yard cover of each cover type. To determine canopy cover, I first measured the yard dimensions using a measuring tape to determine length, from the sidewalk (or street) to the front overhang of the house, and width, from one side of the lot to the beginning of the next lot, including impervious driveways or walkways. Length and width were multiplied to determine total yard area. I then measured the canopy cover of all trees (defined as vegetation taller than 1.5m and excluding cacti) by taking two cross section measurements of each tree's canopy and multiplying these measurements to get a square area. To determine yard canopy cover, I divided the total yard area by the total canopy of trees. In order to capture any biogeochemical impacts from vegetation with N-fixing symbioses (or non-N fixing legumes with high tissue N content, such as palo verde), I also quantified canopy cover for these specific trees. Using the same method as the ground cover, I recorded the vegetative cover of non-trees (referred to as shrub cover, %) by visually segmenting the yard into four quadrants and assigning a % cover of shrubs for each quadrant. I defined shrubs as all plants that were shorter than 1.5m, including cacti and non-annual plants.

2.3 *Rainfall*

Precipitation data were collected via the Flood Control District of Maricopa Country climate sensors (Maricopa County 2015). The climate sensor closest to the study site was used to obtain the precipitation data: ASU South 4525. Data were downloaded for each day of the study period. The precipitation that took place during each deployment was summed for each house, where each house had a unique deployment period and precipitation value.

2.4 *Soil sampling*

To explore soil properties and N pools, I collected soil samples over a two-month period from June to July the summer of 2014 prior to the summer monsoon rains. In each yard, I took four 5 cm diameter cores split into three 15 cm depth intervals to 45 cm using a slide-hammer corer. The 45 cm depth is optimal to capture the effects of plants on nutrient availability because 60-90% of root biomass in aridlands is typically found in the first 40 cm of the soil profile (Reynolds et al. 2004). In turfgrass yards, I took four soil cores by choosing areas of the yard that were at least one meter(m) away from impervious surfaces and a one m away from any tree canopy. Of the four soil cores in each xeric yard, I took two under randomly chosen shrubs <1.5m in height, excluding cacti and annual flowers (hereafter referred to as 'under plant'). I took the remaining two from the vegetation-free patches between shrubs (i.e. places in the yard not covered by canopies of either trees or shrubs, hereafter referred to as 'between plant'). To account for yard level variation I homogenized the two cores within each patch type and depth category (n=6 individual soil samples per yard). For turfgrass yards, I also homogenized within depth category. These samples were bagged and placed in a cooler for 1-2 hours (hrs) until processing in the laboratory.

In the winter of 2014 I sampled soil to determine bulk density using a 5 cm diameter slide-hammer corer as described previously to achieve the least amount of impaction. I took four cores at each site, two in each patch type and at three depths, from 0-45 cm in 15 cm increments. I took special care to collect any soil that may have fallen back into the coring hole due to sandy or loose soils. Samples were bagged and brought back to the laboratory for processing.

2.5 *Ion-exchange resin bags*

I measured N availability in soils using buried ion-exchange resin bag techniques (Giblin et al. 1994) during the summer (July-September) and winter (December-February) rainy seasons as nutrient movement is likely to be maximized during precipitation events. Resin bags were composed of nylon and filled with approximately 10g of 50/50 cation/anion exchange resins (Dowex Marathon MR-3 hydrogen and hydroxide form). I placed four pairs of two resin bags in each yard, two pairs each adjacent to but greater than 30 cm from the place where the ‘between plant’ and ‘under plant’ soil cores were taken. Resin bags were paired to control for small-scale spatial variation from nearby plants or large rocks. One of the pairs in the two-pair set was placed at approximately 5 cm depth and a second bag was placed at approximately 30 cm. For the shallow pair, I used a shovel to lift the soil, then place the paired resin bags under the shovel and then removed the shovel to achieve the least amount of disturbance over the resin bag. To ensure soil sitting above the 30 cm resins was undisturbed and for easy removal and replacement at depth, I augured through the soil at an angle (30-45°) to approximately 30 cm depth and inserted a PVC pipe that held the paired resins on one end (see Appendix A). Resins were kept in the yards for about two months, after which I replaced the resin bags with new resin bags in the holes where they remained for another two-month deployment period.

After each two-month incubation period, I carefully removed the resin bags with a shovel and gloved hands to ensure all nutrients collected by the resins were from the soil. The resin bags were then placed in individual sealed plastic bags and put on ice in a cooler for 1-3 hrs until processing in the laboratory. If processing was postponed more than 3 hrs, resin bags were placed in the refrigerator up to 24 hrs. In the laboratory, I rinsed the resins with deionized water to remove any residual soil. I removed the nylon bag and then each pair of resin bags per hole were homogenized and then dried for 48-72 hrs at 60° C until they stopped loosing weight. After drying, organic materials such as roots, were carefully removed using forceps. Resins were then extracted using 2M sodium chloride (NaCl) solution by shaking for 24 hrs at 160rpms (Giblin et al. 1994). The supernatant was poured through pre-leached Whatman #1 filters and samples were frozen at -4° C until further analyses up to 3 weeks after extraction.

2.6 *Soil Analyses*

I processed soil cores to explore patterns of nutrient pools and rates of microbially mediated N transformation (extractable NO_3^--N and NH_4^+-N content, potential net N mineralization, potential net ammonification, and potential net nitrification) and soil properties related to N cycling (soil moisture, water-holding capacity (WHC), soil organic matter content, texture and pH) All soil methods are based on LTER standard protocols (Robertson et al. 1999). In the laboratory, I sieved the soils using a 2mm sieve within 24 hrs of collection; gravel and organic matter (roots, leaves, insects) were discarded. 20g of sieved soil was set aside at field moisture for both soil moisture determination and WHC, and 10g was weighed out for both exchangeable NO_3^- and NH_4^+ analyses and for incubation to determine microbial N processing rates. I air dried the remaining soil and stored it for future analyses. 20g of soil was dried at 105° C for 24 hrs, and then weighed to determine the moisture content. I determined WHC (100% field capacity of the soils) by saturating 20g of loose soil in a WHC filter funnel using a

Whatman #42 filter and weighing the soil after 24 hrs to determine water content. The 100% WHC value was multiplied by 0.6 to get the 60% WHC which provides soil microbes with both enough water and oxygen in soil pore spaces to optimize microbial processes on dry desert soils (Sponseller 2007). Optimizing microbial activity during the incubations allowed us to measure the greatest potential rates of microbial activity. To determine organic matter content, 10g of oven-dried soils was combusted using a muffle furnace at 550° C for 4 hrs and then weighed again to determine C loss.

To determine pools of soil exchangeable NO_3^- and NH_4^+, I extracted the soils using 100 mL of 2M potassium chloride (KCl) within 48 hrs of collection to retain field conditions. Soil extractions were shaken overnight at 160 rpms and then I filtered the supernatant through pre-leached Whatman #1 filters. To measure microbial N processes, soils were incubated at room temperature (24° C) and in the dark for 7 days at 60% WHC. After the incubation period, I extracted the samples in the same way as the initial soil nutrient samples using a 2M KCl solution. I calculated soil net N mineralization by finding the difference between initial field level NO_3^- and NH_4^+ and nutrient levels at the end of the incubation. After extraction all samples were frozen immediately and thawed when ready to analyze, within 4 weeks.

2.7 *Nutrient Analyses*

I used a colorimetric QuickChem method on a Lachat 8000 FIA system (Lachat Instruments, Loveland, Colorado) to measure NO_3^- and NH_4^+ concentrations in soil extracts. I thawed the samples until they reached room temperature and then shook them before analyses. I corrected the Lachat output data for two blanks, one blank KCl sample and one KCl sample run through a Whatman #1 filter. I also converted these samples from concentrations in mg/L to concentrations in ug of N per g of dry soil using the soil weight (~10g) and extraction volume (100mLs KCl). I diluted any samples that were above detection range (above standard curve, 20 mg/L) using KCl and reran the samples on the Lachat until detection was within the standard curve. I analyzed soils for total carbon (C) and N using a CHN elemental combustion analyzer (PE 2400) after drying and milling at the Goldwater Environmental Lab (Arizona State University).

2.8 *Soil Bulk Density*

To estimate bulk density (BD), I used a hybrid bulk density method recommended by Throop et al. (2012) for rocky aridland soils, where *BD =(mass of fine soil[<2mm])/(core volume -volume of gravel [>2mm]).* Throop's review of BD methods suggests this correction for coarse volume is appropriate when using BD for nutrient analyses pertaining only to the fine fraction because the coarse fraction does not contain extractable inorganic N. Because volumetric BD estimates are inaccurate in rocky soils, I excluded cores in which the gravel volume was greater than 20%. I first calculated BD values for each soil sample, then, because there was no significant difference between patch types, I averaged the BD for each yard across patch type within each depth category. I then averaged across yards, keeping unique BD values for each depth because of significant differences in BD between depth categories (Table 2).

Depth	Coarse + Fine	Fine	(Coarse+fine)-Coarse volume
cm		g/cm^2	
0-15	1.13 (.02)	0.95 (.02)	1.03 (.02)
15-30	0.96 (.03)	0.86 (.03)	0.90 (.03)
30-45	1.19 (.03)	1.09 (.03)	1.15 (.03)

Table 2. Bulk density estimates (g/m^3) for soil samples from 47 residential yards using three different calculation methods (Throop et al. 2012). Values are averages across patch-type because there was not a significant difference in BD between patch types. One standard error (SE) shown in parentheses.

2.9 *Soil Textures*

I determined soil particle size using the hydrometer method (Day 1965). To disperse soil particles I shook 40g of oven-dried soils with 100mL of 50g L (5%) sodium hexametaphosphate for 24 hrs and then quantitatively transferred the soil to a sedimentation cylinder. I added 900mL of deionized water to the cylinder and used a suspension plunger to manually suspend the soil in the cylinder. To determine sand (%) and silt (%) content, I took a hydrometer reading at 40 seconds and at 7 hrs respectively. I calculated clay content using the known percentage of sand and silt subtracted from 100%. To check accuracy of the 40 second sand readings, I sieved the soil after the 7 hr reading using a 53-micron mesh size and determined sand weight by removing remaining sediment from the sieve and drying it overnight at 105° C. The average organic matter content of all samples was 3.5%. According to Gasparotto et al. 2003, not destroying soil organic matter prior to texture analyses decreases precision of silt content by only 1% when using the hydrometer method. Given the low organic matter content in our soils (average 3.5% of xeric, 5.7% for turfgrass) the precision lost for silt values from not destroying the organic matter in the soil texture soils was less than 1%.

2.10 Statistical methods

I conducted all statistical analyses using R 3.0.2 statistical software. I plotted all model residuals to assess assumptions of statistical testing. I corrected soil NO_3^- data for heteroskedasticity using an inverse root transformation (x^(-½)), and resin NO_3^- and NH_4^+ data, organic matter data, C to N (C:N) ratios, and net N mineralization data were log-transformed to achieve homoskedasticity. I ran all tests as mixed model Analysis of Variance (ANOVA) for discrete factors or linear regression for continuous independent variables with house as the random factor. House was designated as a random factor to eliminate non-independence of sub-sampled depths. I also used type III sums of squares for all tests to correct for inconsistent sample size (Winter 2013).

I used ANOVA to assesse both the affect of landscape type (turfgrass and xeric) and year since land cover change (year categories including k=4 groups: <4, 4-8, 8-13, 18-21 years) for soil NO_3^- as a response variable (referred to as "extractable NO_3^-"). To assess patterns of resin NO_3^- (referred to as "plant-available NO_3^-") and NH_4^+ availability I used an Analysis of Covariance (ANCOVA) with precipitation as a covariate to control for effects that were driven by precipitation. Prior to running the ANCOVA, I determined that precipitation and plant-available NO_3^- and NH_4^+ were linearly correlated using a Pearson correlation test.

I then explored the soil properties that may covary or predict soil nutrient patterns between landscape type and over time. To evaluate if organic matter and net mineralization were associated with NO_3^--N I performed 2-way ANOVAs including patch type as a fixed factor. I also used 2-way ANOVAs to test for differences in organic matter and potential net mineralization rates between landscape types and across year category. I then tested for the hypothesized correlation between mineralization and organic matter. To determine if soil texture was influencing plant-available NO_3^- or NH_4^+ I ran an ANCOVA with precipitation as the covariate because nutrient collection by the resin bags directly depends on nutrient movement in water. ANOVA analysis was also used to evaluate texture as a main effect on soil extractable NO_3^- and differences in texture between landscape types. WHC data is not presented here because the capacity of soil to hold water is also measured via sand content and these two variables were highly correlated (Pearson correlation test $p<.0001$). To determine differences in C:N in landscape type and year category, I used two ANCOVAs with shrub cover as the covariate to control for potential C and N inputs (from plant litter and toot turn-over) over time.

To evaluate how multiple management factors influence NO_3- pools simultaneously I performed a mixed linear model with house as a random factor. I used backward stepwise removal to create a model that best predicts xeric extractable NO_3^- levels by including only significant factors and interactions that are ecologically logical. I left all main effects in the model if they were significant or had a significant higher order interaction. I compared second-order Akaike Information Criterion (AIC_c), Delta AICc and Akaike weights between the full factorial model, as well as iterations including bivariate comparisons, to the chosen model to ensure that its likelihood was high compared to other predictive models (Appendix E). Tree canopy cover and N-fixing tree canopy cover variables proved insignificant in the first iterations of the model and were removed thereafter.

To test the relative impact of natural and management-determined variables on xeric yard N cycling, factor analysis was applied to a correlation matrix of soil properties. Factors were extracted using the principal component analyses (PCA) method followed by a varimax rotation (IBM SPSS 21 statistical software). Components with eigenvalues above 1 were retained in the analysis. A variable was said to affect a given component if the factor loading score was above 0.6 or below-0.6. I plotted PCA scores of factors 1 and 2, 2 and 3, and 3 and 4, categorized by years since land cover change, to visually asses any clustering pattern among variables in the model. The PCA helped me to identify which factors account for the most variation in the data

3. Results

3.1 *Soil nitrate pools and availability differ by land cover type and across time*

Soil inorganic extractable-N pools were greater in xeriscapes compared to turfgrass lawns. On average xeriscapes contained 2.5±.4 g NO_3-N/m^2 in the first 45 cm of soil, compared to .6±.7 g NO_3-N/m^2 in lawns (Figure 2, Appendix D). Available NO_3- and NH_4^+ were not significantly higher in xeric yards, but were strongly correlated to precipitation (p<.001, Figure 3, Table 4). Soil extractable NO_3- significantly differed by age category depending on patch type (for interaction Patch-type:Year since land cover change p=.011, Appendix D), while available NO_{3-} was significantly different only between patch types (p=.02). These analyses demonstrate that when there is a change in residential 'grassland' to 'shrubland' cover, there is a significant affect on the availability and pool size of soil NO_3- that varies with time.

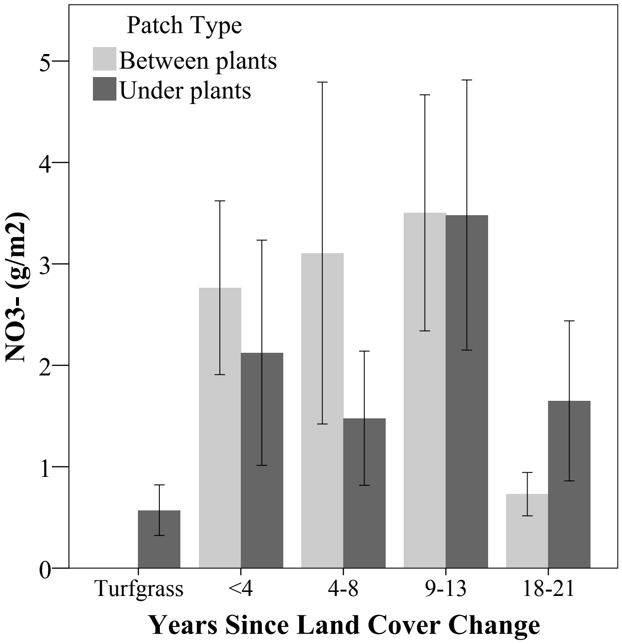

Figure 2. Soil NO_3^--N availability increases after land cover change, specifically in the between plant patch type. Bars represent means of extractable NO_3^- by year since land cover change as a function of patch type, error bars show ± one SE (n=6-12).

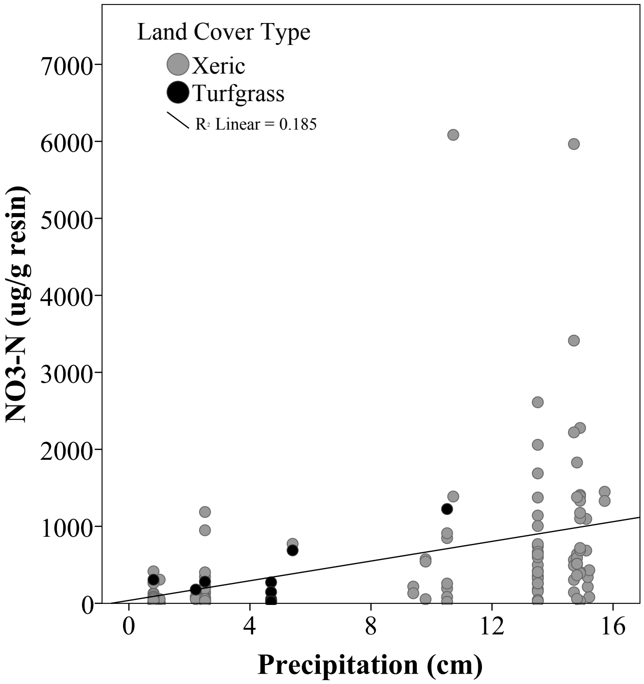

Figure 3. Plant-available NO_3^--N (as estimated from buried mixed exchange resin bags) is significantly related to summer monsoon precipitation received during the resin deployments. Data points represent mean plant-available NO_3^- for two deployments for each study site. Precipitation values, estimated from a nearby NCDC rainfall sensor (ASU South) and are sums of precipitation received during the deployment length for each study site. Deployments took place between June and November of 2014.

	Available NO_3^-	**Available NH_4^+**
	g N/g resin	
Turfgrass	0.64 (.24)	0.08 (.04)
Xeric <4 yrs		
Between Plants	0.57 (.09)	0.12 (.05)
Under Plants	0.65 (.14)	0.14 (.06)
Xeric 4-8 yrs		
Between Plants	1.3 (.41)	0.20 (.08)
Under Plants	1.22(.37)	0.22 (.05)
Xeric 8-13 yrs		
Between Plants	1.25 (.53)	0.15 (.03)
Under Plants	1.09 (.29)	0.47 (.34)
Xeric 18-21		
Between Plants	0.68 (.18)	0.13 (.03)
Under Plants	1.46 (.71)	0.17 (.08)
F value	2.07	0.48
p-value	0.16	0.49

F and p-values from ANOVA of landscape type and $NO3^-$ and NH_4^+

Table 3. Plant-available soil inorganic N means averaged over deployment 1 and 2 (data derived from mixed exchange resins). One SE shown in parentheses.

As predicted, soil properties also show similar patterns as soil nutrients when a turfgrass yard is replaced (Appendix C). Soil organic matter was significantly and negatively related to soil NO_3^--N pools (Pearson's correlation p=.005). Organic matter declined from 6% in turfgrass sites to 4% within 4 years after land cover change, then declined further with time (p=.03 Figure 4A). Organic matter was significantly higher in turfgrass yards then xeric yards (p=.03). I predicted that rates of organic matter loss would be related to rates of N mineralization rates, as organic N is converted to inorganic N by microorganisms. Potential net mineralization rates were not correlated to soil extractable NO_3^--N and did not differ between land cover types or across time. Mineralization rates were however, significantly higher in the under plant patch type than the between plant patch type (p= 0.03) and positively correlated to organic matter (p=.02). Mineralization is not only dependent on organic matter, but also water availability (Leiros 1999). Although there was no significant difference between soil moisture across land cover types, sand content, a measure of the water holding capacity of soils, differed significantly between landscape types (p=.02) with sand content averaging 38% in xeriscapes and 49% in turfgrass (Figure 5, Table 4).

C:N ratios were expected to follow a similar pattern as organic matter and decrease with xeric yard age. C:N ratios significantly decreased over time for all year categories except the oldest category, 18-21, which had higher C:N ratios at each depth than any other year category (Year since land cover change p=.0058; Figure 4B). Contrary to patterns in organic matter, C:N ratios increased with depth (Depth p<.0001, for interaction Year since land cover change:Depth p=.0018).

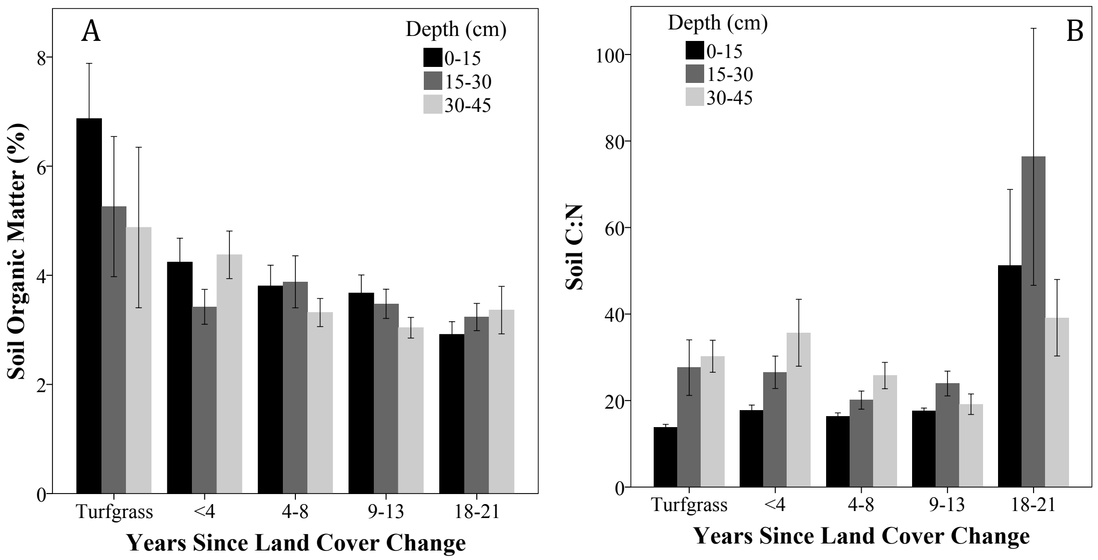

Figure 4. Soil organic matter content decreases across time, while soil C:N vary across landscape type and depth, error bars show ± one SE. A. Soil organic matter content by year since land cover change as a function of soil depth. B. Soil C:N of study sites averaged over patch type and year since land cover change category as a function of depth.

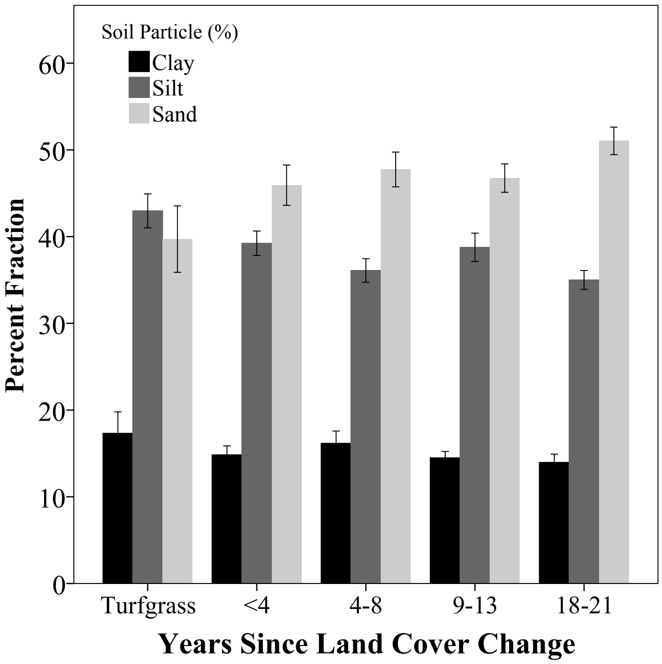

Figure 5. Soil texture shifts from silt and sand dominated in turfgrass yards to sand dominated in xeric yards. Values represent soil texture by particle size obtained via hydrometer method and averaged over both patch type and depth, error bars represent ± one SE.

Table 4. Values represent means averaged over depth of soil analyses (detailed data in appendices). One SE shown in parentheses.

	Gravimetric Soil Moisturet %	Water Holding Capacityt g H_2O/g soil	Soil organic mattert %	Extractable NO_3^--N g/m^2	Extractable NH_4^+-N g/m^2	Sandt %	Soil C:N	Shrub cover %
Turfgrass	**17.45 (2.5)**	**0.63 (.05)**	**5.63 (1.13)**	**0.57 (.25)**	**0.15 (.06)**	**39.71 (3.83)**	**23.50 (2.98)**	**11.73 (4.68)**
Xeric <4 yrsŧ	**9.7 (2.5)**	**0.55 (.05)**	**4.78 (.69)**	**3.10 (.78)**	**0.17 (.04)**	**39.7 (3.83)**	**39.15 (8.84)**	**17.08 (3.5)**
Between Plants	9.89 (1.20)	0.53 (.03)	3.99 (.36)	2.77 (.86)	0.16 (.05)	46.21 (3.46)	25.12 (4.73)	
Under Plants	10.19 (.88)	0.55 (.02)	4.03 (.41)	2.12 (1.11)	0.15 (.03)	45.56 (3.22)	28.66 (4.22)	
Xeric 4-8 yrsŧ	**9.57 (1.46)**	**0.51 (.05)**	**3.67 (.47)**	**3.50 (1.54)**	**0.16 (.05)**	**47.345 (4.78)**	**19.15 (2.37)**	**17.66 (2.45)**
Between Plants	9.07 (1.34)	0.47 (.02)	3.74 (.37)	3.11 (1.69)	0.16 (.03)	48.10 (3.23)	19.07 (2.06)	
Under Plants	9.22 (1.05)	0.48 (.01)	3.67 (.44)	1.48 (.66)	0.12 (.03)	47.39 (2.59)	22.42 (2.00)	
Xeric 8-13 yrsŧ	**9.19 (.1.1)**	**.48 (.02)**	**3.32 (.30)**	**3.5 (1.10)**	**0.14 (.03)**	**46.55 (1.91)**	**25.78 (3.05)**	**8.21 (1.92)**
Between Plants	9.27 (1.1)	0.49 (.02)	3.34 (.31)	3.50 (1.16)	0.14 (.04)	47.49 (2.21)	21.13 (1.70)	
Under Plants	8.19 (.47)	0.50 (.02)	3.48 (.32)	3.48 (1.33)	0.28 (.08)	45.59 (2.53)	19.31 (2.17)	
Xeric 18-21 yrsŧ	**9.1 (1.2)**	**.48 (.02)**	**2.92(.29)**	**.86 (.22)**	**0.14 (.04)**	**51.10 (2.20)**	**35.67 (7.73)**	**15.09 (1.90)**
Between Plants	9.13 (1.27)	0.47 (.02)	2.80 (.29)	0.73 (.21)	0.14 (.05)	50.88 (2.34)	57.49 (18.38)	
Under Plants	8.9 (1.35)	0.49 (.02)	3.58 (.41)	1.65 (.790)	0.21 (.08)	51.19 (2.27)	53.67 (15.41)	
F value	23.41	15.95	12.56	2.16	0.1	5.42	0.45	9.63
p-value	<.001***	<2e-4***	0.001***	0.15	0.75	0.03*	0.51	0.002**

ŧ values in these rows are weighted averages by patch type. Under plant values were weighted by percent area of yard that was under plants. Between plant values were weighted by percent area of yard that was not covered by plants.

t Indicates significant differences between landscape type (p<.05)

* = p<.05 ** =p<.01 ***=p<.001

3.2 Soil nitrate patterns influenced by management over time

Results from the online survey revealed great variety in maintenance of xeric yards and practices used during the land cover change process (see Appendix F). For example, a majority (71%) of homes have drip-irrigation, but over half of the survey respondent indicated they also water using a hose, watering can, or sprinklers. Additionally, about half of survey respondents placed a fabric or plastic barrier on the soil before adding mulch and half indicated removing a tree when converting to a xeriscape.

Patterns in soil extractable NO_3^--N and landscape age were highly influenced by these management practices. The results of the mixed linear model identified several interactions that contributed significantly to the variation in extractable NO_3^--N in xeric landscapes of different age (Table 5). By considering management practices and structure it is evident that extractable NO_3^--N decreased significantly with time. This pattern depends on patch type and is regulated by the amount of irrigation, and whether plastic sheeting is present in the yard. For example, when plastic was present in the between plant patch type, NO_3^--N was lower than when plastic was not present. The inverse pattern is observed in the under plant patch type (Figure 6). Landscapes irrigated more frequently (1+ times per week) contained less NO_3^--N than yards irrigated infrequently (<1 time per week; p=.001), and this relationship was stronger in older xeriscapes than those converted within the past 4 years (Figure 7). Shrub cover also significantly predicted extractable NO_3^--N and this pattern was most evident in the between plant patch type (Figure 8). Results from this model suggest that homeowner decisions help regulate NO_3^--N pool size and that the cover of shrubs may be more important than trees in these landscapes.

Factors & Interactions	denDF	F-value	P-value	
(Intercept)	180	5.49	0.02	
Years since land cover change	41	0.27	0.61	
Shrub cover (%)	41	9.87	<0.01	**
Patch type	180	25.52	<.0001	***
Irrigation frequency	41	1.21	0.28	
Depth (cm)	180	5.12	0.02	**
Presence of plastic	180	0.15	0.70	
Patch type:Depth	180	7.52	<0.01	***
Patch type:Presence of plastic	180	35.08	<.0001	***
Year since land cover change:Patch type	180	13.30	<0.001	***
Shrub cover (%):Patch type	180	12.05	<0.001	***
YearSince:Irrigation frequency	41	1.27	0.27	
Year since land cover change:patch type:Irrigation frequency	180	6.14	0.01	**

**Significant at p<.001 ***Significant at p<.0001

Table 5. Results of mixed model regression. Model was run with study site (House) as a random factor to control for samples taken at multiple depths. All other factors were fixed factors.

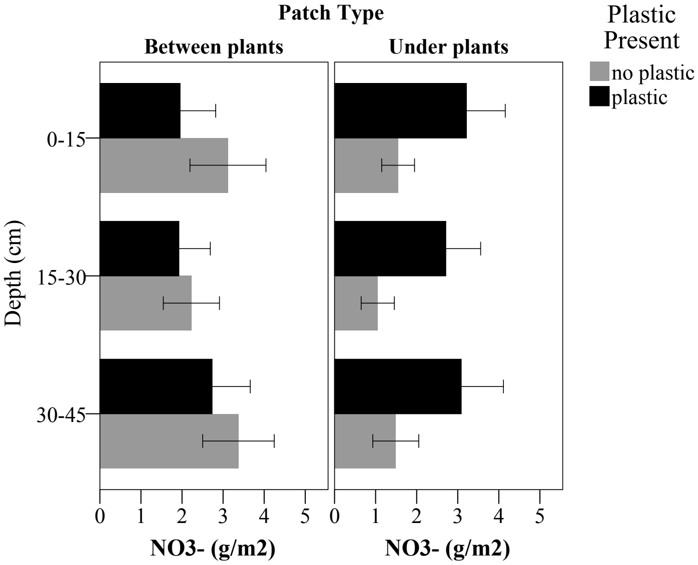

Figure 6. Soil nitrate values are influenced by the presence of plastic sheeting on the soil surface. This interaction is dependent upon patch type, but is not affected by depth of sampling. Bars show average soil extractable NO_3^--N, error bars represent ± one SE.

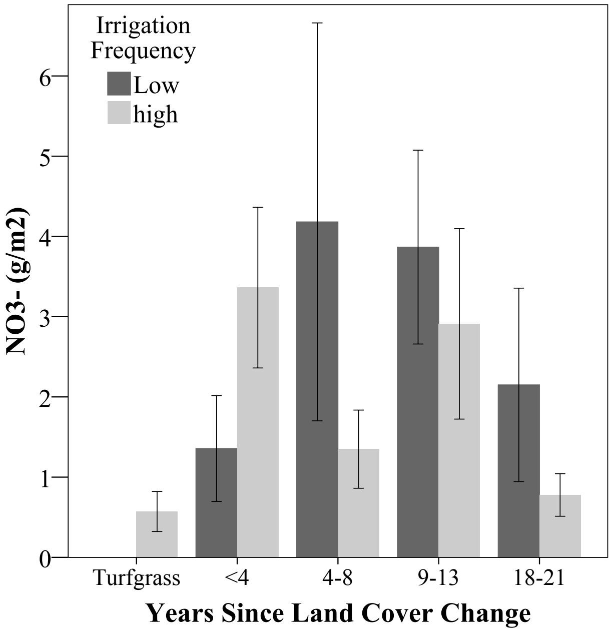

Figure 7. Frequent irrigation decreases nitrate pool sizes compared to infrequent irrigation. High and low irrigation frequency are defined as homeowners that report irrigating at least once a week or less than once per week, respectively. Values are averaged over depth and then weighted by patch type, error bars represent ± one SE.

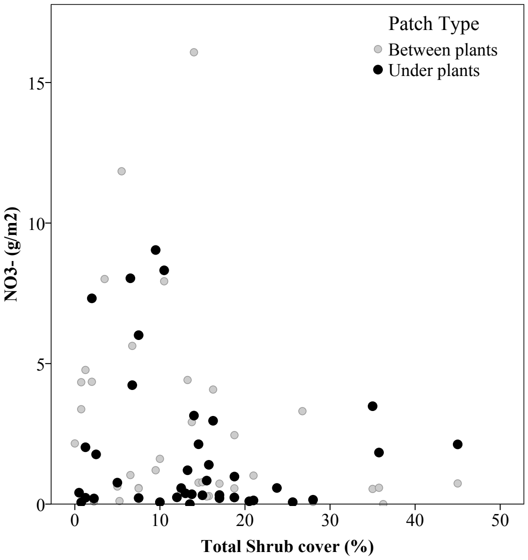

Figure 8. Shrub cover is negatively correlated to soil extractable $NO3^{-}$-N values. Shrub cover is defined as canopy cover of any vegetation under 1.5m in height, excluding annual flowers/weeds.

3.3 Water availability drives N levels in xeriscapes

Results from the PCA analyses reveal that 4 components explained 81% of the variation in these variables (Table 6). The first component was comprised by variables affecting water availability in xeric soils and explained almost 30% of the variation in the data. Vegetation cover, including shrub cover and tree canopy also accounted for approximately 30% of the variation. The principal components scores for all components do not reveal significant patterns between ages since land cover change categories. There is overlap in the turfgrass component scores and the xeriscape scores at age 18-21 years since land cover change (Figure 9).

Table 6. Products of principal component analyses of soil variables show the importance of variables affecting soil water availability. Component one accounted for the highest amount of variance, at 29%. Overall variance accounted fro using PCA factors was 81%.

	Principal Components			
Factors	1	2	3	4
Water-holding capacity	**0.87**	0.01	-0.03	-0.05
% Sand	**-0.81**	0.22	-0.12	0.21
Organic Matter*	**0.76**	0.23	-0.05	0.26
NH_4^+*	-0.06	**0.90**	0.15	-0.09
NO_3^-*	-0.12	**-0.71**	0.46	-0.15
% Shrub Cover	0.05	0.00	**0.95**	0.05
% Canopy Cover	-0.03	-0.01	0.03	**0.97**
Eigenvalue	2.024	1.569	1.073	1.007
Fraction of variance explained (%)	28.91	22.41	15.32	14.39

Extraction Method: Principal Component Analysis. Rotation Method: Varimax with Kaiser Normalization.

a Rotation converged in 5 iterations.

*Indicates the sue of transformed data

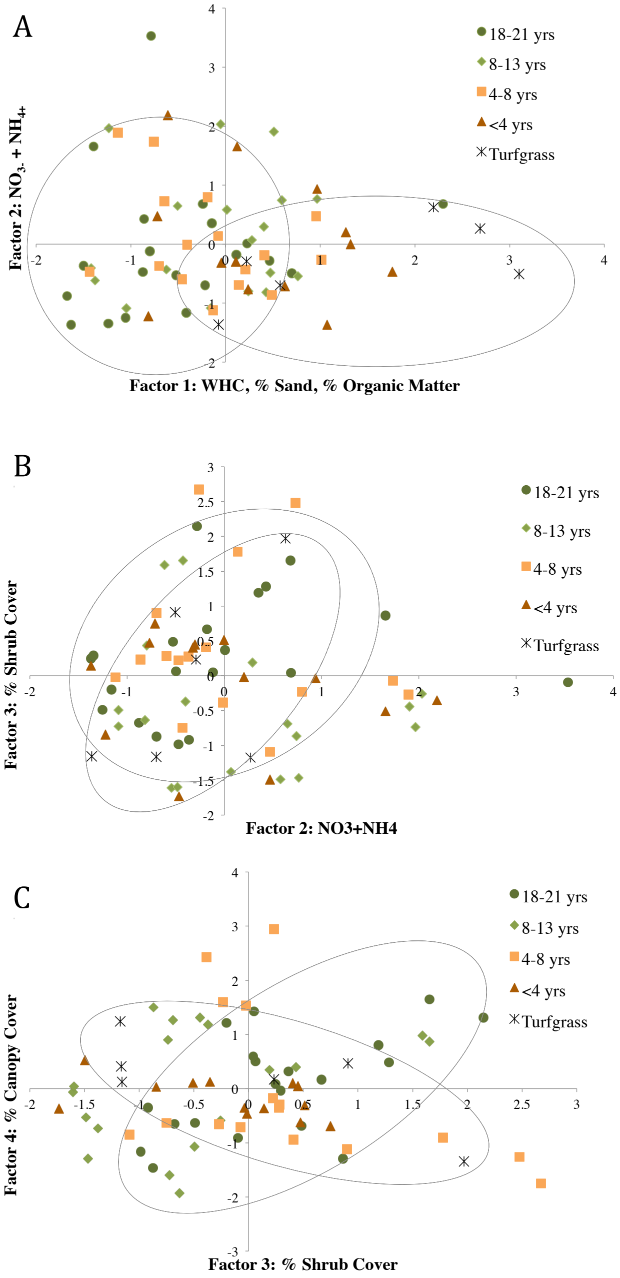

Figure 9. Results of Principal Component Analyses reveal converging of turfgrass landscapes and the oldest grouping of xeric landscapes (18-21 years since land cover change) for factors 2-4. Axes show variation in given components within each Factor: A. Factor 1 and 2; B, Factor 2 and 3; C, factor 3 and 4. Ellipses surround turfgrass values and xeric values in year category 18-21.

4. Discussion

4.1 *Mechanisms determining N availability after land cover change*

I conducted a field study to evaluate the fate of soil inorganic N in residential landscapes after converting from water-intensive turfgrass lawns to ‘sustainable’ xeric yards. After this land cover change there was a significant increase in soil extractable and plant-available NO_3^--N. When monsoonal precipitation occurred, this NO_3^--N was highly mobile in xeric yards compared to turfgrass yards. These finding are in support of previous studies of both natural and constructed ecosystems. Studies have shown that transitioning from grassland to shrubland has implications for water quality due to increased NO_3^--N availability or mobility (Parsons et al. 1996, Baer et al. 2006, Zhu et al. 2006, Turnbull et al. 2010).

Given the high productivity and organic matter turnover of turfgrass landscapes (Pouyat et al. 2006), I hypothesized that the removal of the living grass and replacement with drought-tolerant shrubs would lead to fast decomposition of organic matter resulting in an excess of inorganic N that is not fully utilized by plant uptake of shrubs. Average extractable NO_3^--N levels in turfgrass landscapes were below 6kg/ha, while xeriscapes in the first 13 years after land cover change averaged about 2 kg/ha of NO_3^--N. Organic matter levels dropped within the first 5-8 years after land cover change from an average of 5.7% to less than 3.5%, while NO_3^--N levels peaked in these xeric landscapes. This increase in plant-available N suggests that there is a pulse of decomposition of organic matter within approximately 5 years of turfgrass removal. If organic matter is, in fact, the major source of excess inorganic N in xeric landscapes, there should be a proportional shift in organic to inorganic N between turfgrass and xeric yards as well as decreasing mineralization in the between plant space (Schlesinger et al. 1990). Mineralization rates were significantly higher under plants compared to the between plant space, but there was no significant change across xeric yards of increasing age. Additionally, when the pool-size of inorganic N (from CHN analyses) was extrapolated to whole yard N using site area, results show that on average turfgrass yards had approximately 30% more organic N than the oldest xeric yards. On-the-other-hand, by summing the soil extractable NO_3^--N and NH_4^+-N values and extrapolating to site area, results indicate that xeric yards had approximately 30% more inorganic N than turfgrass yards. These findings support that the decomposition of organic matter is directly resulting in the high amount of plant-available N in xeric landscapes.

The results of this study are in agreement with previous studies on legacy effects of cultivated land (i.e. agriculture) that found soil N accumulation in re-established forests and urban grasslands (Goodale & Aber 2001, Davidson et al. 2007, Raciti et al. 2011). One study of the effects of agriculture on turfgrass C and N determined that despite yard age and varied management by homeowners, soil C pools were significantly higher when yards were previously in agriculture. Lewis et al. (2014) explains that in turfgrass yards, where root turnover is high, decomposition of underlying C pools may never occur as the turfgrass nutrient inputs and uptake are relatively balanced. However, soil N pools in lawns that were previously agriculture, eventually reached similar N levels to lawns that were never in agriculture. This decline of N over time is attributed to both leaching and gaseous losses as well as microbial immobilization. Our data suggest that a similar phenomenon may be occurring when turfgrass is replaced with shrub-dominated alternative landscaping. Nitrate levels in older xeric yards ultimately reach pre-cover change levels, despite high NO_3^--N pool size in young landscapes.

Nitrate levels (in the first 45 cm of soil) could be decreasing in the oldest xeriscapes for several reasons, including nutrient uptake by increasingly mature plants or, in the between plant patch, N-leaching after large rain events. The shift in texture after land cover change from a silt loam dominated soil to a sandy loam xeric soil is likely to lead to increased water infiltration between plants as a result of the higher sand content, causing nutrients to be more mobile at deeper depths (Austin et al. 2004). Studies show that transitioning from grassland to shrubland leads to NO_3^--N leaching below the rooting zone of landscape plants (Bushoven et al. 2000, Amador et al. 2007). Our results from resin bag testing support high concentrations of NO_3^--N in soil solution in xeriscapes (.4mg to >.1mg NO_3^--N/g resin) compared to turfgrass yards (<.4mg NO_3^--N/g resin) but there was no relationship with depth. Our study also found that C:N ratios are much greater in older landscapes. One rationale for the high C:N ratios could be that over time the xeriscapes are leaching dissolved organic N (McLauchlan 2006); this is supported by our findings that silt content decreases with xeric yard age. Another explanation could be low microbial demand for C caused by decreased microbial biomass (Baer et al. 2006).

Wyant et al. (2014), found that both microbial diversity and abundance are significantly lower in xeriscapes than turfgrass yards.

Although I did not evaluate the evaporative force in these landscapes, our results indicate that it may be important in regulating soil moisture and therefore decomposition and mineralization of organic matter in the between plant space in xeriscapes. In SW arid deserts, the evaporative force often leads to shallow infiltration and fast evaporation of surface water from compacted aridisols (Liu et al. 1995, Scanlon et al. 2005). However, soil moisture was not significantly different in xeric yards compared to turfgrass yards, implying that xeric yards do not mirror the ecosystem functioning of native desert ecosystems (Davies & Hall 2010). Additionally, due to the disturbance level and management of residential yards, the soil BD in xeric yards is low compared to desert soils, which may allow deeper water infiltration. When plastic is not present in a xeric yard, NO_3^--N is high in the between plant patch and low in the under plant patch. The high NO_3^--N in the between plant patch could be caused by high evaporation rates that prevent deep infiltration of water. Low NO_3^--N under plants is likely the result of plant uptake when water is added under the plant canopy via drip-irrigation. When xeric yards have plastic sheeting down NO_3^--N levels are high in the under plant patch and low in the between plant patch. This could mean that in the between plant patch the plastic is preventing evaporation, leaving the soil wet for long periods of time. This prolonged moisture could result in denitrification or leaching of NO_3^--N (Paul & Clark 1989, Stevens et al. 1997). In the under plant patch type the plastic may be preventing plant N uptake or this patch type could be receiving localized inputs from plant litter. Despite differences in extractable NO_3^--N pools, there were no significant differences in plant-available NO_3^--N between patch types or any affect of plastic sheeting on plant-available NO_3^--N.

4.2 *Drivers of variation in soil N*

This research not only shows patterns of soil N in different landscape types, it also shows great variation in soil properties, ecosystem function, and ecosystem structure within xeriscapes (Table 2 & Appendix F). The large variance seen in the nutrient and biophysical properties of xeric soils is likely due to the structure and management of these ecosystems. Homeowners make decisions regarding how their yard is converted, how many and what types of vegetation are planted and how the landscape is maintained. For decades lawns have been considered a status symbol, point of pride, and a creative outlet to connect with nature, resulting in great variation in how yards are structured and maintained (Grove et al. 2006, Zhou et al. 2008, Larson et al. 2010). Homeowner decisions determine the magnitude of plant nutrient uptake by controlling the type of vegetation and soil moisture (Lynch 1995, Amador et al. 2007). Differences in vegetation, such as planting native species versus non-natives or shrubs versus cacti, can results in varied root structure. For example, plant allocation of fine root mass tend to be relatively shallow in arid shrub-dominated systems and highly dependent on water availability (Schwinning & Sala 2004). If nutrient-acquiring roots are concentrated at shallow depths they will have a diminished opportunity to intercept plant-available NO_3^--

N deeper in the soil profile (Schenk & Jackson 2002, Amador et al. 2007). Plant root structure is also affected by water availability. Homes that water small amounts more frequently would encourage shallow root growth, while those homes with infrequent large irrigation events, where water infiltration may be higher, could be causing deep rooting to access water several days after an irrigation event (Rundel et al. 1991).

Plant litter is a large source of organic matter in most systems, including forests and grasslands and could be another source of variation in alternative residential landscapes (Moretto et al. 2001, Binkley and Fisher 2012, Fissore et al. 2012). Given the structure of xeriscapes, with rock cover and often plastic or fabric sheeting on the soil surface, it is likely that N inputs from leaf litter or trimmings are variable across sites. In some cases litter may not be in contact with the soil and soil microbes for long enough that it is mineralized. In alternative shrub-dominated landscapes the plant litter is often removed by gathering litter via raking or a leaf blower and then removing by bagging or placement on the street for pickup. In other yards, where rock mulch is over 30 cm thick and there is a plastic barrier down, leaf litter may never reach the soil surface. There are also those yards where rock mulch is sparse and almost all litter may be decomposed and mineralized.

Added water also influences NO_3^--N levels over time. I found that when irrigation was frequent there was less NO_3^--N with time, and while irrigation was infrequent the high NO_3^--N levels persist in older landscapes. There is a decrease in NO_3^--N in the between plant space compared to the under plant space over time, but our data also indicates that in the under plant patch type when irrigation is higher, NO_3^--N is lower. The lower NO_3^--N values observed in the under plant patch type with frequent irrigation could be a result of plant uptake when water is added to the soil in the under plant space. Plants may be taking up nutrients at the surface during irrigation events while nutrients are flushed downward in the between plant space during large precipitation events. In a meta-study of arid pulse dynamics, Collins et al. (2014) supply ample evidence of small precipitation or irrigation events being ample water for microbial mineralization to occur but not enough to trigger plant uptake or growth in arid ecosystems. This pulse-dynamic based theory may apply to some of the xeriscapes in our study, but exclude others. Depending on the structure of the xeriscape, the presence of plastic sheeting, presence of trees, and proximity to turfgrass yards or flood irrigation, some xeriscapes may act similarly to a desert, with patch-specific response to precipitation, while others, those with higher soil moisture, may have more homogeneous response to precipitation events and increased probability of leaching. High irrigation levels could also be causing increasing mineralization rates in the between plant space, leaving larger available N pools that are prone to leaching during monsoon-like rain events. Studies in arid and semiarid environments have consistently found that when soil moisture increases, the mineralization rates also increase, but depending on water inputs and microbial communities the process of mineralizing organic matter from previous land use can take several months to several years (Burke et al. 1997, Bushoven et al. 2000, Austin et al. 2004).

Results of our PCA reveal that despite great variability in the management of these landscapes, three fundamental soil properties account for almost 30% of the variation in the data. Organic matter, WHC, and % sand (Factor 1), are the three variables that determine a. how much NO_3^--N can result from nitrification after decomposition of organic matter, b. the rate of decomposition based on moisture and c. how likely it is for NO_3^--N to leach down the soil column. The turfgrass sites almost all fall on the higher end of the Factor 1 axis and the lower end of the Factor 2 axis, which includes inorganic N. This shows that despite higher affinity for moist soils and more organic matter in turfgrass yards, xeriscapes had higher NO_3^--N.

4.3 Implications & Conclusions

The primary service of alternative landscapes is to save on water cost and consumption. Increasingly common droughts in the Western US have lead to recent legislation to decrease urban water use (DWR 2015). Already, many cities have restrictions on the number of days you can use water for landscaping and a subset of those cities incentivize homeowners and businesses to decrease household water use. Often the restrictions or cutbacks on water use are bottom-up initiatives, starting at the municipal or even grass-roots level. There is likely to be progressively more top-down implementation of statewide regulation of water use. For example, California recently adopted legislation to decrease urban water use by 25% by 2016. Additionally, the US Environmental Protection Agency offers a partnership program, titled WaterSense®, to promote water conservation in communities by providing certification and recommendations for homeowners and businesses with the goal of 20% water use reductions. WaterSense® focuses on water savings in the home and prominently promotes water-wise landscaping (US EPA 2015). Water-wise alternative landscapes are already at the forefront of the initiatives being applied to accomplish water use reduction goals. It is likely that these landscapes will continue to gain popularity across the US.

Our study found that water availability and management determine the pool size and mobility of soil NO_3^--N. For this reason, it is important to consider that more mesic climates, such as California, could see faster decomposition of organic matter and increased concentrations of percolate NO_3^- -N within a shorter time span (Erickson et al. 2001). In order to prevent the accumulation of available NO_3^--N in alternative landscapes and mitigate some potential for negative consequences for water quality due to nutrient leaching, I recommend taking specific steps during the process of residential land cover change. Completely removing grass using a turf cutter would eliminate a large source of organic matter in new landscapes. I also suggest that amendments to soil, such as fertilizer or compost be added only where plants roots will reach to avoid unnecessary mineralization in unplanted patches of mulch. Placing a plastic barrier used to be common practice to prevent weeds and regrowth of grass. Currently it is recommended to place weed-blocking fabric after the removal of grass and then mulch on top of this to allow for evaporation in the between plant patches. To slow the decomposition process by avoiding trapping moisture under the mulch it would be beneficial to allow the soil to dry prior to putting down fabric and mulch. Planting annual native flowers may contribute to plant uptake and would require minimal water inputs. These steps may improve nutrient retention in alternative landscapes, but more research is needed across a precipitation gradient and on a variety of alternative landscape types in order to determine potential trade-offs between water conservation and water quality.

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APPENDIX A

INSTALLING ION-EXCHANGE RESIN BAGS

Figure A1. The 30cm depth resins were installed using a double-PVC pipe apparatus. The outside larger pipe was permanently installed at an angle in the soil during the study. The smaller, inner pipe, was used to hold and replace the resins. The resins sat at the bottom end of the tube (shown here upside down) and the opposite end was capped using plastic sheeting to prevent any precipitation or irrigation water reaching the resin bags that did not first travel through the soil column.

APPENDIX B

VARIATION IN XERIC YARDS


KUBY








Figure B1. Photos of select study sites show great variation in yard structure, maintenance, and plant composition.

APPENDIX C

ANOVA AND ANCOVA TESTING

Independent Variables		NO3- (g/m2)	Resin NO3-(ug/g resin)	Resin NH4+ (ug/g resin)	Organic Matter (%)
Landscape Type	df	44	41	41	42
	f	**2.16**	**2.07**	**0.48**	**12.56***
Years since land cover change	df	38	36	41	42
	f	**2.84**	**0.14**	**1.02**	**11.23**
Patch Type	df	175	207	207	
	f	**8.36***	**1.21**	**5.2**	
Shrub cover (%)	df	38	36	36	
	f	**7.03***	**0.15**	**1.19**	
Sand (%)	df	163	207	207	
	f	**1.95**	**0.23**	**0.83**	
Significant Interactions & Covariates		Shrub: Location Shrub: Year Category	Precipiation	Precipitation	

Table C1. Results of ANOVA and ANCOVA testing. Dependent variables are represented in column titles with independent variables as row titles. Starred values indicate significant differences (alpha=.05). All ANOVA tests were run with study site (House) as a random factor to control for non-independence of pseudo-replicates.

APPENDIX D

AVAILABLE AND EXTRACTABLE NO_{3-} DATA BY DEPTH

	Gravimetric Soil Moisture	**Water Holding Capacity**	**Soil organic matter**	**NO3-N**	**NH4+**
	%	g H20/g soil	%	g/m2	g/m2
Xeric yards	0.09 (.003)	0.50 (.01)	3.53 (.10)	2.38 (.23)	0.17 (.01)
Between plants	0.09 (.004)	0.50 (.01)	3.41 (.12)	2.61 (.35)	0.15 (.01)
0-15	0.08(.007)	0.48 (.02)	3.38 (.22)	2.62 (.65)	0.19 (.030
15-30	0.10 (.007)	0.51 (.01)	3.48 (.23)	2.10 (.50)	0.11 (.03)
30-45	0.10 (.006)	0.49 (.01)	3.36 (.20)	3.11 (.63)	0.15 (.01)
Under plants	0.09 (.004)	0.50 (.01)	3.67 (.16)	2.09 (.29)	0.19 (.02)
0-15	0.08 (.006)	0.50 (.01)	3.88 (.28)	2.28 (.48)	0.16 (.03)
15-30	0.10 (.007)	0.52 (.01)	3.51 (.23)	1.74 (.46)	0.20 (.06)
30-45	0.09 (.005)	0.50 (.01)	3.63 (.32)	2.25 (.57)	0.20 (.05)
Turfgrass yards	0.18 (.017)	0.63 (.03)	5.72 (.71)	0.58 (.16)	0.15 (.05)
Under plants	0.178 (.02)	0.63 (.03)	5.72 (.71)	0.58 (.16)	0.15 (.12)
0-15	0.20 (.031)	0.67 (.05)	6.87 (1.01)	0.85 (.37)	0.21 (.12)
15-30	0.18 (.037)	0.64 (.07)	5.26 (1.28)	0.56 (.25)	0.08 (.02)
30-45	0.15 (.015)	0.58 (.05)	4.89 (1.47)	0.29 (.18)	0.16 (.06)

Table D1. Averages of both xeric and turfgrass soil properties by detpth, one SE shown in parentheses.

	NH4+	**NO3+NO2**
	ug N/g resin	
Xeric	82.87(11.8)	515.71(81.6)
Between plant	83.55(11.5)	520.06(79.1)
shallow	96.12(188)	397.07(73.8)
deep	69.82(15.2)	632.67(144.2)
Under Plant	128.62(36.6)	581.33(110.2)
shallow	127(21.3)	452.16(72.9)
deep	151.83(78.6)	799.13(233.4)
Mesic	16.44(3.7)	168.06(42.8)
shallow	17.78(5.6)	204.86(64.1)
deep	14.42(4.7)	112.85(42.8)

Table D2. Averages of both xeric and turfgrass soil available NO_{3-} and NH_4^+ sites by. One SE shown in parentheses.

APPENDIX E

RESULTS OF MIXED LINEAR REGRESSION MODEL SELECTION

Table E1. Mixed linear model selection using logical backward stepwise removal revealed important factors controlling soil extractable NO_{3-} in xeric yards. Shows selection criterion for model possibilities. AICc was used do to low sample size. Model 6 had the lowest deltaAICc and highest weight, indicating that it is a model that explains the data well. I added NO_{3-} significant ecological interactions to this simplified model, shown in model 7.

Model: All model include House as a random factor and use a Restricted Maximum Likelihood criterion	**AIC**	**AICc**	**dAICc**	**Weight** [AICc]
rootno3~YearSince*total.Shrub..*Location*IrrigationFrequency*Depth*PlasticPresent	472.600	525.558	750.543	>0.000
rootno3~YearSince+total.Shrub..*Location*IrrigationFrequency*Depth*PlasticPresent	39.892	52.619	277.604	>0.000
rootno3~YearSince+total.Shrub..+Location*IrrigationFrequency*Depth*PlasticPresent	-163.054	-159.110	65.874	>0.000
rootno3~YearSince+total.Shrub..+Location+IrrigationFrequency*Depth*PlasticPresent	-187.958	-186.303	38.682	>0.000
rootno3~YearSince+total.Shrub..+Location+IrrigationFrequency+Depth*PlasticPresent	-214.522	-213.535	11.449	0.003
rootno3~YearSince+total.Shrub..+Location+IrrigationFrequency+Depth+PlasticPresent	-225.789	-224.985	0.000	0.851
rootno3~YearSince+total.Shrub..+Location+IrrigationFrequency+Depth+PlasticPresent+(Depth:Location)+(Location:PlasticPresent)+(YearSince:Location)+(total.Shrub..:Location)+(YearSince:IrrigationFrequency)+(YearSince:Location:IrrigationFrequency)	-223.669	-221.467	3.518	0.147

APPENDIX F

RESULTS OF ONLINE SOCIAL SURVEY

Survey Question Topic	Survey Question	% of Yards
Conversion process	To the best of your knowledge, when your yard was changed to a desert-style landscape, how was the yard prepared?	
	Stopped watering grass	45.2
	Living grass removed (removed grass while still alive)	41.9
	Stopped water and used herbicide	38.7
	Used turf removal equipment (removed first 5cm soil)	22.6
	Kept living grass (installed new cover over living grass)	2
	Other	22.6
Soil amendments during conversion	Before adding rocks or other non-grass ground cover, which of the following activities were completed?	
	Laid down barrier (plastic sheeting or weed-blocking fabric)	22.6
	Tilled soil	16.1
	Added soil or compost	3.2
	Watered grass or dirt	0
	Added fertilizer	0
	Other	9.7
Litter management	Which of the following best describes what is done after maintaining the yard?	
	Cut grass, weeds, leaves/branched and remove by bagging	92.9
	Leaf-blower is used to blow off leaves, weeds, or grass	28.6
	Grass clippings, leaves or weeds are left on the ground	9.5
	Grass, shrubs or weeds are not cut or trimmed	2.4
	Don't know	2.4
Irrigation frequency	Thinking about this past summer (June-August 2013), about how often / were the plants, trees or grass in your yard usually watered?	
	Every day	4.8
	Every week (but not daily)	54.8
	Every month (but not weekly)	21.4
	Every season (but not monthly)	4.8
	Never water	11.9
Irrigation type	Please indicate all of the ways used to water plants, trees, or grass in your yard.	
	Drip irrigation	71.4
	A handheld hose	52.4
	Watering jug/can	35.7
	Sprinklers	31
	Sprinklers on a hose	14.3
	Flood irrigation	4.8
	Other	71

Table F1. Descriptive statistics on xeric yard management and the land cover change process collected from a 2014 social survey of homeowner who participated in the Tempe Landscape Rebate Program.