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Physical and biological controls on trace gas fluxes in semi-arid urban ephemeral waterways

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Abstract Rapid increases in human population and land transformation in arid and semi-arid regions are altering water, carbon (C) and nitrogen (N) cycles, yet little is known about how urban ephemeral stream channels in these regions affect biogeochemistry and trace gas fluxes. To address these knowledge gaps, we measured carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) before and after soil wetting in 16 ephemeral stream channels that vary in soil texture and organic matter in Tucson, AZ. Fluxes of CO_2 and N_2O immediately following wetting were among the highest ever published (up to 1,588 mg C m^{-2} h^{-1} and 3,121 μg N m^{-2} h^{-1}). Mean post-wetting CO_2 and N_2O fluxes were significantly higher in the loam and sandy loam channels (286 and 194 mg C m^{-2} h^{-1}; 168 and 187 μg N m^{-2} h^{-1}) than in the sand channels (45 mg C m^{-2} h^{-1} and 7 μg N m^{-2} h^{-1}). Factor analyses show that the effect of soil moisture, soil C and soil N on trace gas fluxes varied with soil texture. In the coarser sandy sites, trace gas fluxes were primarily controlled by soil moisture via physical displacement of soil gases and by organic soil C and N limitations on biotic processes. In the finer sandy loam sites trace gas fluxes and N-processing were primarily limited by soil moisture, soil organic C and soil N resources. In the loam sites, finer soil texture and higher soil organic C and N enhance soil moisture retention allowing for more biologically favorable antecedent conditions. Variable redox states appeared to develop in the finer textured soils resulting in wide ranging trace gas flux rates following wetting. These findings indicate that urban ephemeral channels are biogeochemical hotspots that can have a profound impact on urban C and N biogeochemical cycling

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pathways and subsequently alter the quality of localized water resources.

Keywords Urban stream · Trace gas flux · Semi-arid · Nitrogen · Carbon · Methane

Introduction

Arid and semi-arid ecosystems are experiencing disproportionate increases in human population and land transformation worldwide resulting in altered hydrosystems and biogeochemical cycles (Ezcurra 2006; Kaye et al. 2006; Grimm et al. 2008a). Numerous studies have focused on how urbanization alters the redistribution of water, nutrients and biogeochemical responses in upland arid and semi-arid ecosystems (Kaye et al. 2006; Kennedy 2007; McCrackin et al. 2008; Hall et al. 2009), as well as the consequences for runoff and the delivery of nutrients to areas of focused groundwater recharge (Lewis and Grimm 2007; Grimm et al. 2008a; Gallo et al. 2012a, 2013). However, little is known about how the morphology of urban streams, specifically ephemeral streams that dominate arid and semi-arid regions (Levick et al. 2008), alter N transformations and removal via trace gas fluxes and what controls those fluxes. Understanding water and materials fluxes in urban environments and the role that ephemeral streams play in controlling trace gas fluxes will be critical for developing science-based management strategies for the long term sustainability of urban ecosystems in water limited environments (Grimm et al. 2008a; Goddard et al. 2010). This is especially true in the rapidly urbanizing semi-arid Southwestern United States (Grimm et al. 2008a, b) where urban runoff is actively managed to augment a limited water supply, and reducing delivery of nitrate to focused areas of recharge is critical to sustaining groundwater quality (Carlson et al. 2011).

Recent research in the rapidly urbanizing Southwestern US indicates accelerated carbon (C) and nitrogen (N) cycling and gas losses in upland ecosystems owing to changes in the redistribution of water, carbon and nitrogen resulting from land use (Fenn et al. 2003; Lohse et al. 2008). Lohse et al. (2008), for example, show that urbanization in the Central Arizona Phoenix Long-term Ecological Research (CAP LTER) study area increases C and N loading to urban core ecosystems via dry and wet deposition compared to desert ecosystems; and Hall et al. (2008) measured higher rates of nitrogen cycling and nitric oxide and nitrous oxide losses from grass lawns compared to xeric landscaped areas. Other studies show that downstream engineered lake and grass catch basins have higher C and N-cycling rates and thus higher potential for N loss via denitrification than xeric gravel basins (Zhu et al. 2005; Larson 2010; Roach and Grimm 2011).

Despite the prevalence of ephemeral streams (81 % of southwestern streams, Levick et al. 2008) and a projected doubling of urban land cover by 2,050 in the semi-arid southwest (Theobald et al. 2013), to our knowledge, no studies have measured trace gas losses in ephemeral stream channels, especially those undergoing urbanization. Findings from Gallo et al. (2012a, b, 2013) in Tucson, AZ indicate that drainage networks and the characteristics of urban ephemeral streams, such as substrate texture, play important roles in controlling nutrient load patterns in urban runoff and soil solute loads following a runoff event. These findings suggest that much of the nutrient processing and C and N gas flux occur within ephemeral channels between wetting events, subsequently altering urban runoff quality. Research in upland ecosystems show immediate biogeochemical and biological responses to rainfall following periods of soil water limitation (Cable and Huxman 2004; Reynolds et al. 2004; Schwinning and Sala 2004). In arid and semi-arid regions, rainfall pulses can sustain soil biological processes (Snyder and Williams 2000; Loik et al. 2004; Belnap et al. 2005) which under warmer spring and summer temperatures, can increase microbial activity and responses such as trace gas fluxes (Conant et al. 2004; Bowling et al. 2011).

The objective of this research was to examine how stream channel characteristics control summertime trace gas losses in response to soil wetting following an extensive dry period in ephemeral urban streams, or washes, in Tucson, Arizona, USA. We focus on summertime trace gas fluxes when warm temperatures and summer rainfall may enhance biogeochemical processes. We selected 16 sites in ephemeral stream channels that ranged in soil texture and organic matter and measured trace gas fluxes prior to and following soil wetting. We used linear regression and nonparametric statistical analyses to quantify relationships between site characteristics and flux and identify likely mechanisms controlling gas flux responses.

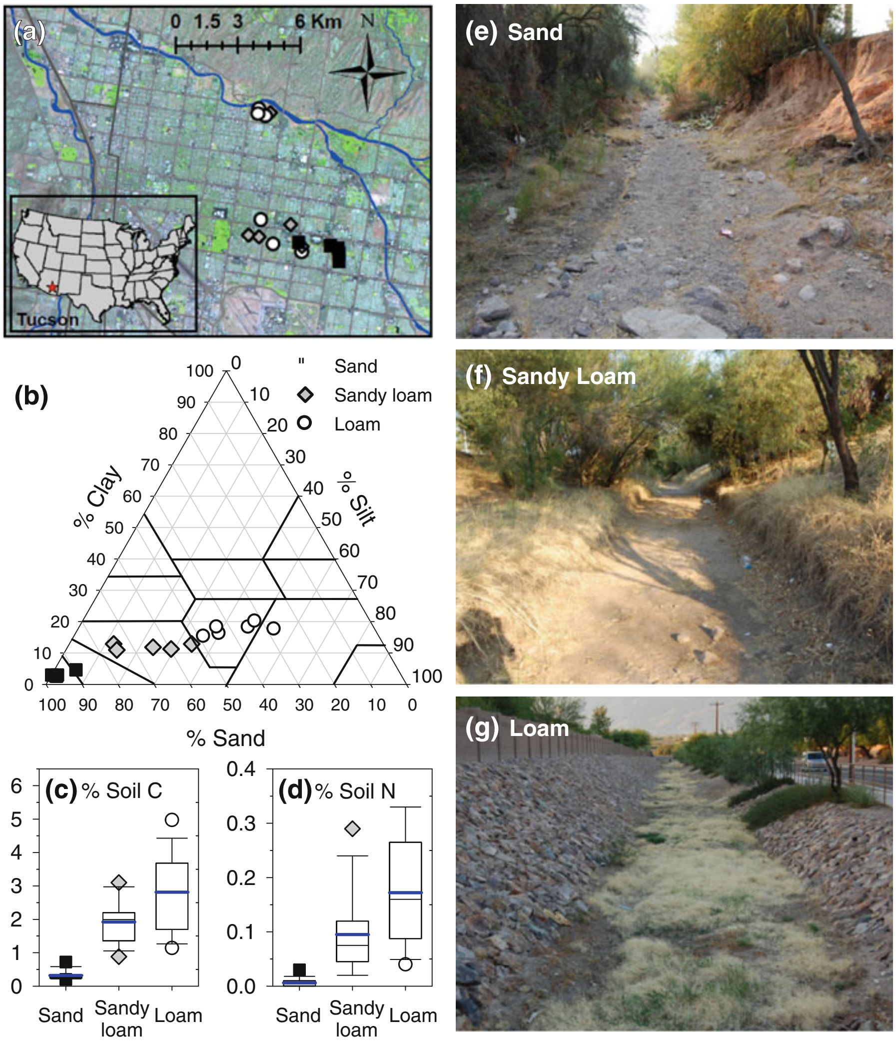


Fig. 1 **a** Location of urban ephemeral washes in Tucson, Arizona included in this study, **b** soil texture properties of the study washes, percent soil **c** carbon (C) and **d** nitrogen (N) used to classify the study sites into 3 major categories: **e** sand washes, **f** sandy loam washes and **g** loam washes. The thick *blue box plot lines* indicate means, the thin *black lines* indicate medians. (Color figure online)

Study site description

This study was conducted in ephemeral washes across Tucson, Arizona (Fig. 1a) prior to the first summertime rainfall event, following an extensive period of dry conditions (3–4 months). The climate is semi-arid with a mean annual rainfall of 310 mm, mean annual evaporation of 270 mm and potential evaporation of 1,900 mm. Rainfall occurs during the summer and winter with summertime atmospheric convection resulting in intense rainfall of short duration (The North American Monsoon, Guido 2008), whereas moisture systems originating in the Pacific Ocean generate cool, protracted winter rainfall.

Stream flow in these urban Tucson washes is ephemeral and occurs only in response to rainfall (Gallo et al. 2012a, 2013). Similar to non-urban streams, significant stream channel transmission losses result in waterways that might experience repeated wetting due to rainfall during the summertime months, but that exhibit a small number of seasonal runoff events (Houser et al. 2000; Gallo 2011; Zhang et al. 2011). The majority of summertime rainfall events in this region tend to be less than 20 mm in magnitude (Cable and Huxman 2004; Loik et al. 2004; Bowling et al. 2011), with the average rainfall event at these sites being between 8 and 11 mm (Gallo et al. 2013).

Table 1 Physical characteristics of the study ephemeral channels

Site	Class	Soil texture (%)[a]			Bulk density (g cm^{-3})[b]	Fraction of fine earth[c]	% Soil organic matter[d]	% Soil C[e]	% Soil N[e]
		Sand	Silt	Clay					
1	Sand	96.9	0.1	3.0	1.6 (0.04)	0.41 (0.02)	0.4 (<0.1)	0.2 (<0.1)	<0.1 (<0.1)
2	Sand	95.8	1.6	2.6	1.67 (0.24)	0.50 (0.06)	0.2 (<0.1)	0.3 (<0.1)	<0.1 (<0.1)
3	Sand	96.0	1.3	2.7	1.86 (0.05)	0.52 (0.01)	0.4 (< 0.1)	0.3 (< 0.1)	<0.1 (<0.1)
4	Sand	95.3	1.7	3.0	1.91 (0.08)	0.60 (0.04)	0.5 (<0.1)	0.3 (<0.1)	<0.1 (<0.1)
5	Sand	89.4	6.0	4.6	1.9 (0.07)	0.72 (0.03)	0.5 (0.1)	0.6 (0.1)	<0.1 (<0.1)
6	Sandy loam	74.8	12.3	13.0	1.69 (0.11)	0.80 (0.01)	2.6 (0.5)	1.8 (0.3)	0.1 (<0.1)
7	Sandy loam	59.7	29.0	11.3	1.68 (0.25)	0.63 (0.06)	1.3 (0.1)	1.3 (0.1)	<0.1 (<0.1)
8	Sandy loam	74.9	14.1	11.0	1.32 (0.21)	0.74 (0.02)	2.2 (0.2)	1.8 (0.1)	0.1 (<0.1)
9	Sandy loam	64.4	23.8	11.8	1.09 (0.24)	0.89 (0.06)	2.7 (0.8)	2.0 (0.6)	0.1 (<0.1)
10	Sandy loam	53.2	34.0	12.9	1.38 (0.12)	0.86 (0.05)	6.3 (1.6)	2.5 (0.3)	0.2 (<0.1)
11	Loam	34.8	47.0	18.2	1.24 (0.23)	0.90 (0.05)	6.8 (1)	3.5 (0.1)	0.2 (<0.1)
12	Loam	48.7	36.0	15.3	1.21 (0.06)	0.77 (0.06)	2.5 (0.7)	1.5 (0.3)	0.1 (<0.1)
13	Loam	43.5	38.1	18.3	1.03 (0.12)	0.72 (0.05)	4.5 (0.1)	2.7 (0.1)	0.2 (<0.1)
14	Loam	27.9	54.5	17.6	0.73 (0.09)	0.94 (0.04)	7.5 (2.3)	3.6 (0.7)	0.2 (0.1)
15	Loam	43.9	39.9	16.2	1.21 (0.09)	0.95 (0.01)	2.3 (<0.1)	1.6 (0.1)	0.1 (<0.1)
16	Loam	32.0	47.8	20.2	0.85 (0.04)	0.98 (0.01)	8.5 (0.1)	4.0 (0.2)	0.3 (<0.1)

Values reported are means (±SE; n = 3) with the exception of soil texture where n = 1 and final soil moisture and water filled pore space which were modeled using Hydrus 1-D. Superscripts indicate the methodology used to characterize each physical or chemical soil parameter

[a] Soil texture (% sand, % silt and % clay) was determined using the modified pipette method for particle size analyses (Gee and Bauder 1986)

[b] Bulk density (g cm^{-3}) was determined using a modified version of the excavation method (Grossman and Reinsch 2002)

[c] The fraction of fine earth was calculated as one minus the rock volume divided by the sample volume

[d] % soil organic matter was assumed to be equal to the mass lost on ignition (Gallo et al. 2012b)

[e] Percent soil carbon (% soil C) and soil nitrogen (% soil N) were determined on an Elemental Analyzer (ThermoElectron Corporation Bremen, Germany) at Idaho State University

We selected 16 locations in ephemeral stream reaches spanning a range of soil textures and soil C and N (Table 1; Fig. 1b–d). Based on soil texture, the study sites were grouped into 1 of 3 ephemeral channel substrate classes: (1) sandy, (2) sandy loam and (3) loam (Fig. 1e–g). Average sand content was 94.7 ± 1.3 (standard error, SE), 65.4 ± 4.2 and 38.5 ± 3.3 % in the sand, sandy loam and loam washes, respectively. Average silt + clay content was 5.3 ± 1.3, 34.0 ± 4.2 and 61.5 ± 3.3 % in the sand, sandy loam and loam washes, respectively. Bulk density was highest in the sand (1.60–1.91 g cm^{-3}) followed by sandy loam (1.09–1.69 g cm^{-3}) and loam washes (0.73–1.24 g cm^{-3}). Percent SOM, determined by mass loss-on-ignition, was on average 0.4 ± <0.1, 3.0 ± 0.9 and 5.4 ± 1.1 % in the sand, sandy loam and loam washes, respectively. Percent soil C was on average 0.3 ± 0.1, 1.9 ± 0.2 and 2.8 ± 0.4 % in the sand, sandy loam and loam washes, respectively, while % soil N was on average <0.1 ± <0.1, 0.1 ± <0.01 and 0.2 ± <0.1 % in the sand, sandy loam and loam washes, respectively (Table 1).

Methods

Experimental design for trace gas sampling

We conducted an artificial rainfall experiment and monitored trace gas fluxes prior to and after soil wetting following protocols described by Hall et al. (2008). In brief, gas fluxes were monitored using static gas chambers consisting of a polyvinyl chloride (PVC) chamber base and cap. Chamber bases were installed at least 1 h prior to any gas monitoring, and trace gases were sampled before (t = −1), immediately after (t = 0) and for several hours (t = 0.5, 2 and 6 h) after wetting. We evenly wetted the soil within the static chamber with an artificial rainstorm of depth of 10 mm applied over 15 min at a rate of 40 mm h^{-1}. This rainfall depth and intensity is well within the range of observations made elsewhere in the region (Mendez et al. 2003; Cable and Huxman 2004; Bowling et al. 2011) and within urban Tucson catchments (Gallo et al. 2013). Soil wetting was performed at 8 am across all our sites in order to facilitate the execution of the experiment and to ensure that field work was completed during daylight hours. We capped the chamber base and collected five gas samples, one every 15 min to calculate the gas flux rate for each monitoring period (t = −1, 0.5, 2 and 6 h after wetting) using linear regression of gas concentration inside the chamber versus time. Gas samples were collected using a 20 ml syringe and 10 ml glass Wheaton bottle with inert grey butyl rubber crimp caps. Bottles were vented with needle and flushed with 20 ml of gas sample and then injected with a second 20 ml sample. At the end of each monitoring period (t = −1, 0.5, 2 and 6 h after wetting), the gas chamber cap was removed and the air inside the chamber was allowed to equilibrate with the atmosphere. The first gas sample collected following soil wetting was considered the instantaneous abiotic gas flux (t = 0). Soil temperature was monitored during every sampling period using a soil thermometer.

The gas samples were analyzed for carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) on a gas chromatograph (GC) fitted with a thermal conductivity detector (TCD), an electron capture detector (ECD), and flame ionization detector (FID) (Agilent Technologies, Santa Clara, CA), respectively, in the Hall Laboratory at Arizona State University. The GC was calibrated using certified CO_2, N_2O, and CH_4 standards (Scott Specialty and Matheson Tri-Gas). We used temperature and barometric pressure data from AZMET (http://ag.arizona.edu/azmet/) and the chamber volume to determine the mass of CO_2 as C in mg and N_2O as N and CH_4 as C in µg during each monitoring event. Gas flux rates in mg m^{-2} h^{-1} for CO_2 as C, µg $m^{-2}h^{-1}$ for N_2O as N and CH_4 as C during each monitoring period (time = −1, 0.5, 2 and 6 h since wetting) were calculated via linear regression of gas concentrations versus number of minutes elapsed since the chamber cap placement and divided by the chamber area. Total or cumulative CO_2, N_2O and CH_4 flux over the monitoring period was calculated by integrating the flux rates over time from the onset of wetting to the end of the experiment.

Physical and biogeochemical processes explaining fluxes

To minimize disturbance to the soils and chambers, soils were only collected after the wetting experiment and analyzed for soil water content, N pools, and N cycling. Triplicate soil cores (0–5 cm deep) were collected adjacent to the gas monitoring area and brought back to the lab for immediate processing. Soils were sieved to 2 mm and initial gravimetric soil moisture (initial % soil moisture) was determined by drying a 25 g subsample at 105 °C; and we calculated the percent water filled pore space (%WFPS) by dividing soil moisture by the total porosity and multiplying by 100 (Sarrantonio et al. 1996). Nitrogen pools and transformation rates were measured using aerobic laboratory incubations on composite soil samples from each of the study reaches following methods modified from Hart et al. (1994). In brief, we extracted inorganic N from a soil subsample with 2 N potassium chloride (KCl, 1:5 soil to extract ratio) by shaking it on an orbital mixer for 1 h and then filtering it through a pre-leached Whatman 1 filter. Another subsample was incubated in the dark at room temperature (~25 °C) for 7 days after which inorganic N was extracted as described above. Soil extracts were analyzed for nitrite + nitrate nitrogen (NO_2+NO_3 as N, NO_3 henceforth) and ammonium nitrogen (NH_4 as N) in a SmartChem Discrete Analyzer (Westco Scientific Instruments, Brookfield, CT, limit of detection = 0.001).

We used HYDRUS-1D (Šimunek et al. 2005) to estimate % soil moisture and %WFPS over time at each of our sites, and estimate final % soil moisture and %WFPS. We used % sand, % silt, % clay and bulk density (Table 1) to parameterize the water characteristic curve at each site, and used evaporation flux data from AZMET (http://ag.arizona.edu/azmet/) to constrain evaporative soil water losses.

Statistical analyses

All statistical analyses were carried out with JMP 10.0.2 statistical software (SAS, Cary, NC). We used the Wilcoxon non-parametric comparison of means (Zar

1999) to identify significant ($p \leq 0.05$) differences among the substrate groups on bulk density, % soil moisture, fraction of fine earth, %WFPS, % SOM, % soil C, % soil N, pre-wetting flux rates and cumulative CO_2, N_2O and CH_4 fluxes; soil inorganic N (NH_4–N and NO_3–N + NO_2–N), net mineralization and net nitrification. In addition, we used Wilcoxon non-parametric comparison of means to test for differences in pre-wetting trace gas flux rates (t = −1) from instantaneous (t = 0) and post-wetting (t ≥ 0.5) gas flux rates. We used linear regression to determine if flux rates in each soil category varied in response to soil temperature, % soil moisture, %WFPS, and to determine if cumulative flux, net mineralization and net nitrification varied in response to bulk density, initial and final % soil moisture, fraction of fine earth, initial and final %WFPS, % SOM, % soil C, % soil N to cumulative flux. We used factor analysis (Kaiser 1958; DeCoster 1998; Lehman et al. 2005) to identify patterns and potential mechanisms controlling stream channel biogeochemistry across all sites and within each soil category. Flux rates, net mineralization and nitrification, and all the aforementioned soil characteristics were included in these analyses. Only factors explaining more than 10 % of the data variance were retained for further analyses and interpretation. Variables with loadings greater than 0.70 < −0.70 were considered to heavily load onto that factor and were retained for analyses interpretation.

Finally, to put into a larger context the trace gas flux rates measured here, we compare our results to fluxes quantified in a number of studies in semi-arid, arid and humid climates in both, urbanized and non-developed landscapes. We report published values including flux rate mean and rate minimum and maximum. We have standardized all rates to mass of C (mg) or N (µg) h^{-1} m^{-2}.

Results

Soil temperature and moisture

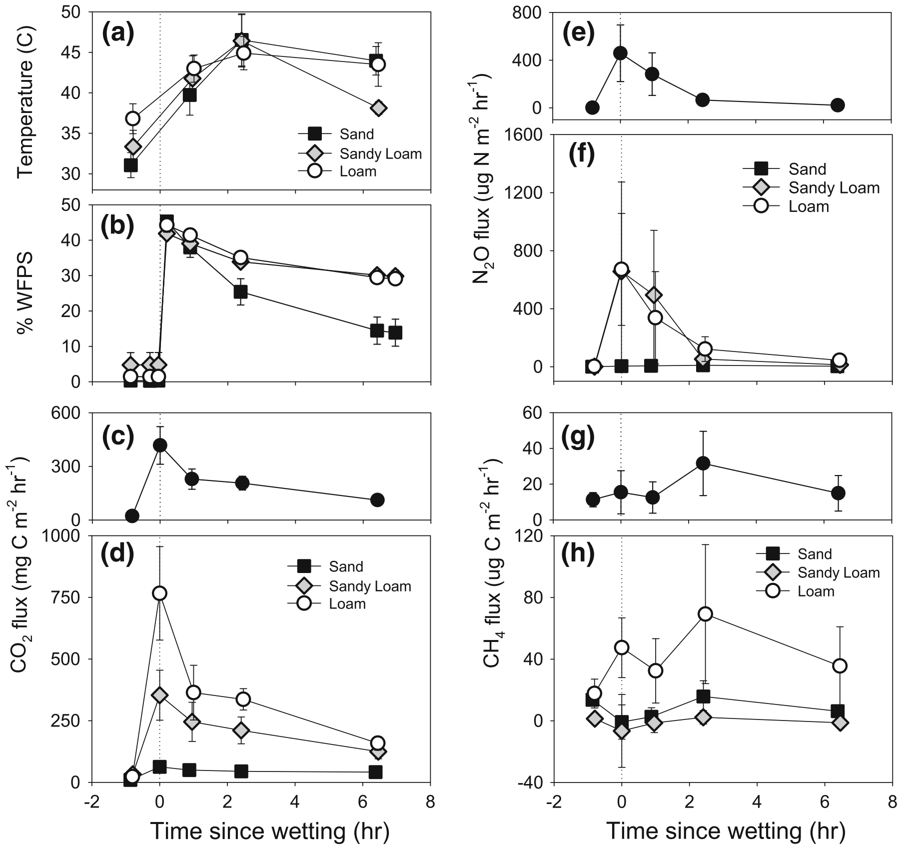
Soil temperatures were lowest in the morning (34.7 °C ± 0.5 SE) before soil wetting, were highest at mid-day, 2 h after soil wetting (45.7 °C ± 0.6, Fig. 2a). Initial soil moisture was 0.1 ± <0.1 % in the sandy sites, 1.2 ± 0.7 % in the sandy loam sites and 0.9 ± 0.3 % in the loam washes. Initial %WFPS was 0.4 ± 0.2 % in the sand washes, 4.8 ± 3.5 % in the sandy loam and 1.5 ± 0.6 % in the loam washes. Hydrus soil moisture modeling indicated that immediately after wetting (t = 0), % soil moisture and %WFPS across sites were 17.7 ± 3.1 and 43.9 ± 1.0 %, respectively and decreased over time thereafter (Fig. 2b). At the end of the experiment % soil moisture was significantly higher in the loam (18.5 ± 3.6 %) than in the sandy loam washes (9.9 ± 1.5 %), and in the sandy loam washes than in the sand washes (2.3 ± 0.5 %). Significant differences in final %WFPS occurred only between the loam and sand washes (29.1 ± 1.2 and 13.9 ± 3.9 %, respectively).

Trace gas fluxes and nitrogen processing

Fluxes of CO_2 and N_2O increased following soil wetting across sites and substrates (Fig. 2c–g). Pre-wetting fluxes of CO_2 were significantly ($p < 0.05$) higher in the sandy loam and loam sites than in the sand sites (Table 2), and across sites, CO_2 fluxes significantly increased after wetting from 21.9 ± 4.2 (SE) mg C m^{-2} h^{-1} to 240 ± 34.0 (SE) mg m^{-2} h^{-1}. Instantaneous (t = 0) CO_2 fluxes across sites were surprisingly large (417.2 ± 105.42 mg C m^{-2} h^{-1}), significantly higher than fluxes at t = 0.5, 2 and 6 h (Fig. 2c), and significantly higher and more variable in the loam that in the sand sites (Fig. 2d; Table 2). Fluxes of CO_2 remained low and unchanged in the sand sites following wetting, whereas flux rates decreased over time in the sandy loam and loam sites (Fig. 2d).

Similarly, post-wetting N_2O fluxes (207.4 ± 76.3 µg N m^{-2} h^{-1}) were significantly higher than pre-wetting N_2O fluxes (1.5 ± 0.7 µg N m^{-2} h^{-1}, Fig. 2e) and were significantly higher and more variable in the sandy loam and loam sites than in the sand sites (Fig. 2f; Table 2). Instantaneous N_2O fluxes across sites were also surprisingly large (458.6 ± 237.7 µg N m^{-2} h^{-1}) and significantly higher in the loam than in the sand sites (Table 2). Fluxes of N_2O remained elevated following wetting at the sandy loam and loam sites, and decreased significantly over time at the loam sites (Fig. 2f). Unlike CO_2 and N_2O fluxes, there were no significant differences or temporal patterns in pre-wetting, instantaneous or post-wetting CH_4 fluxes across sites (Fig. 2g), which averaged 17.1 ± 5.07 µg C m^{-2} h^{-1}. However, post-wetting CH_4 fluxes were significantly

Fig. 2 Variation in **a** soil temperature, **b** percent water filled pore space (%WFPS), fluxes of CO_2 carbon **c** across sites and **d** within each soil texture group, fluxes of N_2O nitrogen **e** across sites and **f** within each soil texture group, fluxes of CH_4 carbon **g** across sites and **h** within each soil texture group over the duration of the 6 h monitoring period including pre-wetting fluxes. *Symbols* denote means (±SE)



higher in the sand and loam that in the sandy loams sites, and most variable at the loam sites (Table 2; Fig. 2h).

Cumulative CO_2 losses over the monitoring period were significantly greater in the loam than in the sandy loam sites (Fig. 3a; Table 2); and although N_2O and CH_4 losses were not significantly different across substrates (Fig. 3b, c), cumulative N_2O losses were highly variable in the sandy loam (SE = 789 mg C m^{-2}) and loam sites (SE = 742 μg N m^{-2}), while CH_4 losses were highly variable in the loam sites (SE = 157 mg C m^{-2}).

Initial soil NH_4 and $NO_2 + NO_3$ were significantly higher in the loam than in the sand or sandy loam washes (Fig. 4a, b; Table 2). Average net mineralization and nitrification rates were not significantly different across substrate categories, due primarily to highly variable responses in sandy loam and loam sites where both net production and net consumption occurred (Table 2; Fig. 4c, d).

Correlations and factor analyses

Fluxes of CO_2 varied positively and significantly with soil temperature at the sand and sandy loam washes (coefficient of determination, $r^2 = 0.38$ and 0.27, respectively), and across sites with soil moisture conditions, with the strongest correlations being those of CO_2 flux versus % soil moisture at the sandy loam sites ($r^2 = 0.62$) and %WFPS at the sand and loam sites ($r^2 = 0.47$ and 0.41, respectively). The only significant N_2O flux correlation observed was of a significant N_2O flux increase with %WFPS ($r^2 = 0.17$) at the sand sites. Similarly, the only significant correlation observed for CH_4 fluxes was a positive one with soil temperature ($r^2 = 0.25$) at the loam sites only.

Cumulative CO_2 flux varied significantly with soil texture and associated physical and biogeochemical variables across sites. Specifically, cumulative CO_2 fluxes exhibited significant linear, positive correlations with % clay ($r^2 = 0.58$), % silt ($r^2 = 0.59$), the

Table 2 Mean (±SD) pre-wetting, instantaneous and post-wetting trace gas flux rates, and cumulative flux during the experimental period, initial pools of soil NH_4 and NO_3 and net mineralization and nitrification rates for each of the soil texture groups

	Sand	Sandy loam	Loam
CO_2 flux (mg C m^{-2} h^{-1})			
Pre-wetting	9.8 (2)^b	32.5 (26.4)^a	23 (4.5)^a
Instantaneous	62.6 (40.6)^b	353.2 (226.9)^{ab}	765.9 (463.9)^a
Post-wetting	45.4 (33)^b	193.5 (131.1)^a	286.2 (187.3)^a
Cumulative flux (mg m^{-2})	224.9 (220.8)^b	1019.4 (677)^{ab}	1691.1 (729)^a
N_2O flux (ug N m^{-2} h^{-1})			
Pre-wetting	0.6 (0.8)^a	0.9 (1.5)^a	2.7 (4.4)^a
Instantaneous	4.9 (6.1)^b	656.9 (1378.5)^{ab}	671.4 (944.4)^a
Post-wetting	6.9 (12.6)^b	187.1 (579.9)^a	168.1 (457.3)^a
Cumulative flux (ug m^{-2})	43 (85.4)^a	973.2 (1765.4)^a	957.2 (1817.2)^a
CH^4 flux (μg C m^{-2} h^{-1})			
Pre-wetting	13.6 (6.2)^a	1.4 (8.1)^b	17.7 (23)^{ab}
Instantaneous	−0.8 (24.9)^a	−6.6 (52.9)^a	47.4 (47.5)^a
Post-wetting	8.2 (15.3)^a	−0.1 (9.5)^b	45.7 (76.1)^a
Cumulative flux (μg m^{-2})	−27.2 (79.2)^a	−7.1 (14.6)^a	201.4 (385.4)^a
Initial NH_4 (μg N g^{-1} soil)	0.9 (1.1)^b	1.6 (1.1)^b	5.5 (1.5)^a
Initial NO_3 (μg N g^{-1} soil)	6.8 (11.6)^b	26.2 (31.2)^b	177 (126.2)^a
Net mineralization (μg N g^{-1} day^{-1})	0 (0.1)^a	0.6 (1.3)^a	4.7 (12.8)^a
Net nitrification (μg N g^{-1} day^{-1})	0.1 (0.2)^a	0.8 (1.4)^a	4.9 (11.7)^a

Means sharing the same superscripted letter across columns indicate values that are not significantly ($p > 0.05$) different

fraction of fine earth ($r^2 = 0.63$), % SOM ($r^2 = 0.61$), % soil C ($r^2 = 0.68$) and N ($r^2 = 0.61$), soil NH_4 ($r^2 = 0.49$) and NO_2+NO_3 ($r^2 = 0.39$), and final soil moisture ($r^2 = 0.38$). Meanwhile, cumulative CH_4 fluxes, net mineralization and net nitrification varied positively and significantly with soil NH_4 ($r^2 = 0.31$) and NO_2+NO_3 ($r^2 = 0.65$). In contrast, cumulative N_2O fluxes were not significantly correlated to any measured soil variables.

Because many of these site-specific soil characteristics covary, factor analysis provides a quantitative assessment of these joint variations and the resultant relationships with observed gas fluxes. Factor analyses including data from all sites did not identify any general relationships between soil characteristics and trace gas responses. In contrast however, factory analyses within each soil texture class showed clear relationships between gas fluxes and substrate physical and chemical characteristics. In the sand washes, 3 factors accounted for 95.3 % of the data variability, with most of the variance explained by Factor 1 (54.2 %), with cumulative CO_2 and N_2O flux, soil NO_3, the fraction of fine earth, initial and final soil moisture and %WFPS, and % soil C and N loading positively onto Factor 1. Net mineralization, net nitrification and soil NH_4 loaded positively onto Factor 2, while pre-wetting CH_4 flux loaded heavily and negatively onto Factor 2, which explained 22.7 % of the data variance (Fig. 5a). Pre-wetting N_2O flux and bulk density loaded positively, while cumulative CH_4 flux and pre-wetting CO_2 flux loaded negatively onto Factor 3, which explained 18.4 % of the data variance (Table 3).

At the sandy loam washes, 2 factors accounted for 80.9 % of the data variance (Table 3). Additional factors did not increase individual factor loadings or improve the interpretation of the factor analyses. Variables loading on Factor 1 were % SOM, % soil C and N, soil NH_4 and NO_3, net nitrification and mineralization, and cumulative N_2O and CH_4 flux, and explained 46.6 % of the data variance. Factor 2 explained 34.3 % of the data variance with fraction of fine earth, post-monitoring % soil moisture and %WFPS, and cumulative CO_2 flux loading positively on Factor 2, and initial % soil moisture and %WFPS loading negatively (Fig. 5b).

Finally, at the loam washes, Factors 1, 2, 3 and 4 explained 29.1, 28.3, 21.8 and 15.7 % of the data variance, respectively (94.9 % total, Table 3). Cumulative CH_4 flux, pre CH_4 flux, net mineralization, net nitrification and soil NO_3 loaded positively onto Factor 1, while pre-wetting N_2O loaded negatively.

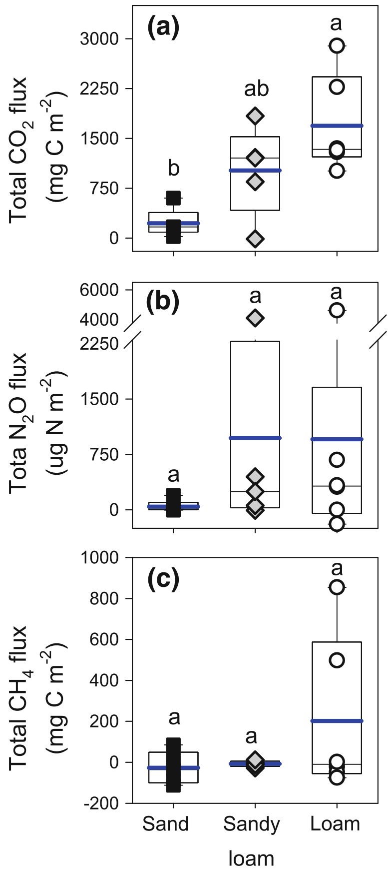


Fig. 3 Total cumulative fluxes (±SE) over the 6 h of the experiment as **a** carbon dioxide (CO_2), **b** nitrous oxide (N_2O), and **c** methane (CH_4) for sand, sandy loam, and loam substrates. The thick *blue boxplot lines* indicate means, the *thin black lines* indicate medians. *Box plots* not sharing the same *letter* are significantly ($p < 0.05$) different. (Color figure online)

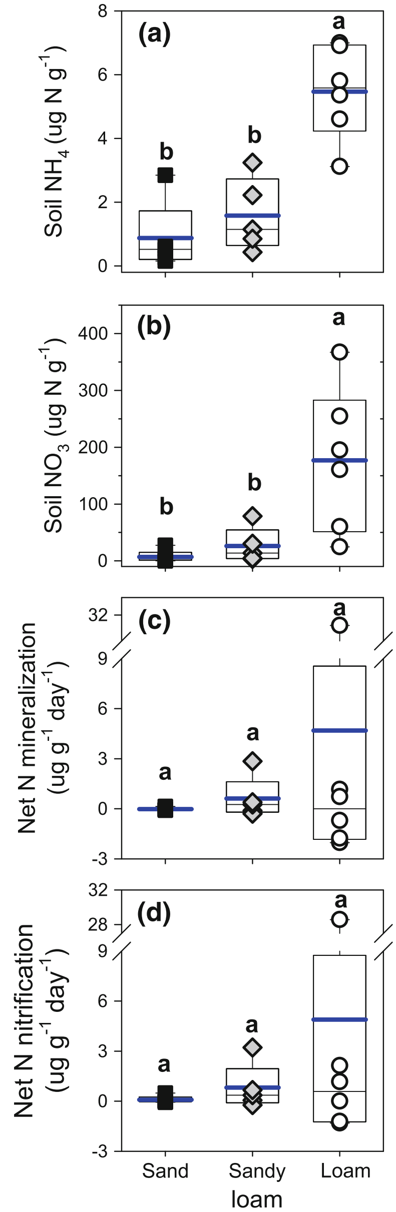


Fig. 4 Initial **a** soil ammonium-N (NH_4), **b** nitrate–N (NO_3), **c** net mineralization and **d** net nitrification for the sand, sandy loam and loam substrates. The *thick blue boxplot* lines indicate means, the *thin black lines* indicate medians. *Box plots* not sharing the same letter are significantly different ($p < 0.05$). (Color figure online)

Final % soil moisture, %WFPS, % SOM and % soil C and N loaded positively onto Factor 2, while bulk density loaded negatively (Fig. 5c). Cumulative CO_2 flux, initial % soil moisture and %WFPS loaded positively onto Factor 3 while soil NH_4 and pre CO_2 flux loaded positively and cumulative N_2O flux loaded negatively onto Factor 4 (Fig. 5d).

Discussion

Few studies have examined how urbanization in water limited regions alters fluxes of C and N gases in hotspots such as ephemeral streams, although it is well established that urbanization alters C and N loading and nutrient cycling pathways (Pouyat et al. 2002; Kaye et al. 2006; Grimm et al. 2008a; Lorenz and Lal 2009; Pataki et al. 2011). Our results demonstrate that trace gas fluxes from these washes are among the largest gas fluxes observed (e.g. Sponseller 2007; Groffman and Pouyat 2009; Townsend-Small et al.

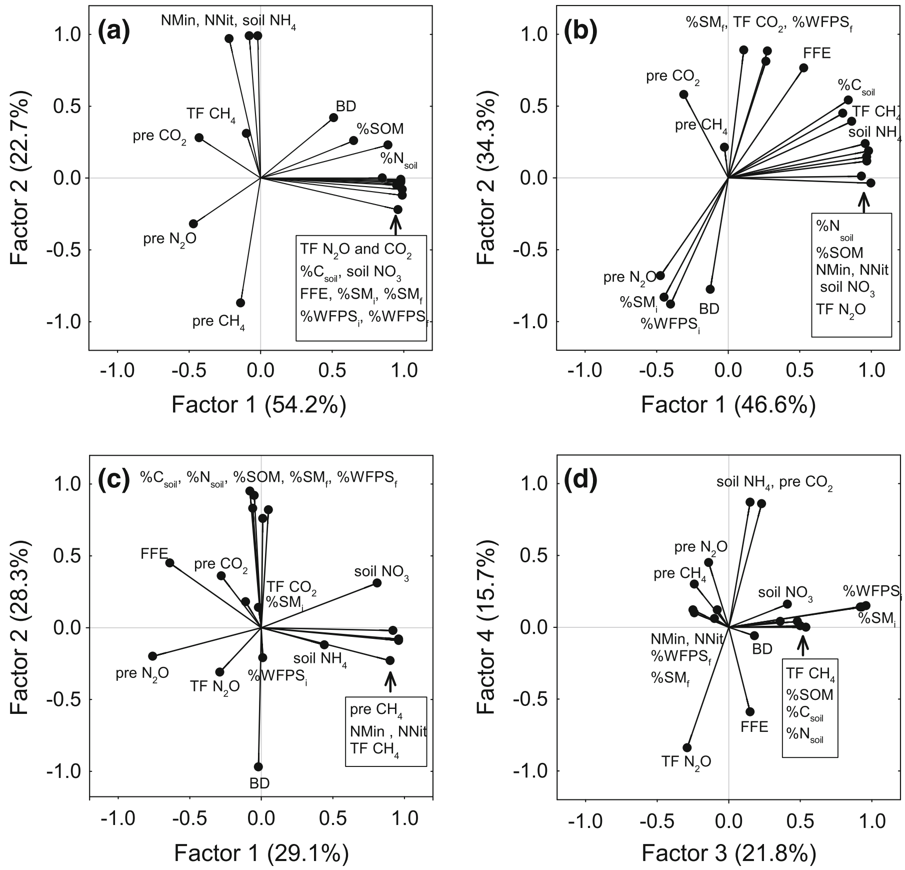


Fig. 5 Variable loadings onto factors 1 and 2 at the **a** sandy **b** sandy loam and **c** loam sites, and **d** factors 3 and 4 at the loam sites

"pre" indicates pre-wetting CO_2, N_2O and CH_4 fluxes.
TF CO_2, TF N_2O, TF CH_4 indicate total (cumulative) trace gas flux.
$\%SM_i$ and $\%SM_f$ are percent soil moisture initial and final, respectively.
$\%WFPS_i$ and $\%WFPS_f$ are water filled pore space initial and final, respectively.
BD, FFE are bulk density and fraction of fine earth, respectively.

2011, Table 4) and that the magnitude, duration, and direction of these fluxes varies based on the stream channel substrate characteristics. These urban streams are areas of focused runoff that are disproportionately important in regulating water quality and potential nutrient loading to groundwater (Gallo et al. 2012a, b, 2013). The extremely high instantaneous fluxes we observe following wetting are indicative of physical displacement of soil gases that accumulated over time (Marañón-Jiménez et al. 2011; Kim et al. 2012) while sustained high fluxes over the 6 h observation period indicate rapid biological response to added moisture (e.g. McLain et al. 2008; Unger et al. 2010). Together, these data suggest that ephemeral channels are both hot spots of biogeochemical cycling on the landscape that also act as biogeochemical hot moments following wetting (McClain et al. 2003; Harms and Grimm 2008).

Temporal patterns in trace gas fluxes

Not surprisingly, in a strongly water limited system all trace gas fluxes were low prior to wetting (Huxman et al. 2004; Belnap et al. 2005; Harms and Grimm 2012). Our pre-wetting CO_2 fluxes were lower than fluxes observed in desert uplands and in more humid non-urban systems (Table 4, McLain and Martens 2006; McCrackin et al. 2008), but were similar to those observed in a more humid metropolitan area (Table 4, Raciti et al. 2011). Given the dry pre-wetting conditions in this study, it is likely that pre-wetting CO_2 fluxes are driven in part by photodegradation of

Table 3 Factor analysis summary

	Sand			Sandy loam		Loam			
	Factor 1	Factor 2	Factor 3	Factor 1	Factor 2	Factor 1	Factor 2	Factor 3	Factor 4
Variance explained (%)	54.2	22.7	18.4	46.6	34.3	29.1	28.3	21.8	15.7
Cumulative variance (%)	54.2	76.9	95.3	46.6	80.9	29.1	57.4	79.2	94.9
Factor loadings									
Cumulative flux									
CO_2 (mg C m^{-2})	**0.98**	−0.01	0.17	0.27	**0.88**	−0.11	0.18	**0.93**	0.14
N_2O (μg N m^{-2})	**0.99**	−0.08	−0.05	**1.00**	−0.04	−0.29	−0.31	−0.29	**−0.84**
CH_4– (μg C m^{-2})	−0.10	0.31	**−0.76**	**0.80**	0.45	**0.90**	−0.23	0.36	0.04
N−processing rates (μg N g^{-1} soil day^{-1})									
Net mineralization	−0.22	**0.97**	0.01	**0.97**	0.12	**0.96**	−0.08	−0.24	0.10
Net nitrification	−0.08	**0.99**	−0.06	**0.97**	0.15	**0.96**	−0.09	−0.25	0.12
Pre-wetting fluxes									
CO_2 (mg C m^{-2} h^{-1})	−0.43	0.28	**−0.86**	−0.31	0.58	−0.28	0.36	0.23	**0.86**
CH_4 (μg C m^{-2} h^{-1})	−0.14	**−0.87**	0.39	−0.03	0.21	**0.92**	−0.02	−0.24	0.30
N_2O (μg N m^{-2} h^{-1})	−0.47	−0.32	**0.82**	−0.47	−0.68	**−0.76**	−0.2	−0.14	0.45
Bulk density (g/mL)	0.51	0.42	**0.75**	−0.13	**−0.78**	−0.02	**−0.97**	0.18	−0.06
Fraction of fine earth	**0.85**	0	0.53	0.53	**0.76**	−0.64	0.45	0.15	−0.59
Initial soil moisture (%)	**0.96**	−0.04	−0.12	−0.45	**−0.83**	−0.02	0.14	**0.92**	0.14
Initial WFPS (%)	**0.98**	−0.02	−0.1	−0.40	**−0.88**	0.01	−0.21	**0.96**	0.15
Final soil moisture (%)	**0.96**	−0.22	0.11	0.11	**0.89**	−0.08	**0.95**	−0.10	0.06
Final WFPS (%)	**0.95**	−0.05	0.30	0.26	**0.81**	−0.05	**0.92**	−0.08	0.12
Soil organic matter (%)	0.65	0.26	0.42	**0.98**	0.19	−0.06	**0.83**	0.48	0.04
% soil C	**0.98**	−0.03	0.15	**0.84**	0.54	0.05	**0.82**	0.50	0.01
% soil N	**0.89**	0.23	0.35	**0.96**	0.24	0.01	**0.76**	0.54	0
Soil NH_4 (mg N g^{-1} soil)	−0.02	**0.99**	−0.16	**0.86**	0.39	0.44	−0.12	0.15	**0.87**
Soil NO_3 (mg N g^{-1} soil)	**0.99**	−0.12	0.02	**0.93**	0.01	**0.81**	0.31	0.41	0.16

Loading values for variables that heavily load (variable loading >0.70 or <−0.70) onto a factor are noted in bold

WFPS water filled pore space

organic matter (Rutledge et al. 2010), particularly at the sandy sites where soils were driest. Pre-wetting N_2O and CH_4 fluxes were similar to those observed in bare soils of undeveloped deserts (Billings et al. 2002; McLain et al. 2002, 2008), suggesting that in addition to soil moisture limitations, pre-wetting fluxes of CO_2 are also limited by soil C resources. Large, initial trace gas fluxes following wetting are consistent with previous work in arid and semi-arid regions (Austin et al. 2004; Cable and Huxman 2004; Huxman et al. 2004; Sponseller 2007; Sponseller and Fisher 2008). However, our instantaneous CO_2 and N_2O fluxes are among the highest ever published (Fig. 2d, f; Table 2) and are most likely indicative of physical displacement of gases that have accumulated within the soil following previous wetting events (Marañón-Jiménez et al. 2011; Kim et al. 2012). High variability in these instantaneous fluxes may be due to site differences in antecedent microbial activity, or possibly rapid increases in microbial metabolism (Sponseller 2007). Indeed, high fluxes following soil wetting have been described as the “Birch Effect”, a set of biogeochemical responses to a wetting pulse following a period of drought that include enhanced decomposition of labile soil organic matter, increased rates of nitrogen mineralization (Birch 1958) and high post-wetting CO_2 flux (Jarvis et al. 2007; McLain et al. 2008; Unger et al. 2010; Kim et al. 2012; Navarro-García et al. 2012).

Given the extremely dry soils it is likely that these fluxes result primarily from physical displacement of soil gasses, but further work is needed to evaluate the potential contribution of immediate microbial responses.

Surprisingly, temperature did not emerge as a major control on trace gas fluxes at these sites. Although the soil temperatures measured here are consistent with diel temperature fluctuations observed in other regional studies (McLain and Martens 2006; Hall et al. 2008), temperature explained a relatively small fraction of the trace gas flux patterns we observed, including 38 and 27 % of the CO_2 fluxes at the sand and sandy loam sites and 25 % of CH_4 fluxes at the loam sites. Consistent with the results of Conant et al. (2004), the effect of soil temperature on trace gas fluxes appears to be overshadowed by the release of early summer soil water limitations in these urban ephemeral streams, which facilitate the physical displacement soil gasses and stimulate microbial activity.

Magnitude of trace gas fluxes relative to previous studies

Overall, Fluxes of CO_2 and N_2O from the sandy loam and loam washes were almost 30 times higher and more variable than in the sand washes, and overall higher than those measured in both, humid and water limited urban and non-urban environments (Table 4, Kaye et al. 2004; McLain and Martens 2006; Allaire et al. 2008; McCrackin et al. 2008; Groffman et al. 2009; Raciti et al. 2011); with the exception of CO_2 fluxes, which were similar to those in a non-urban ephemeral stream (Table 4, Sponseller 2007). Indeed, N_2O fluxes at the sandy loam and loam washes were as much as 2 orders of magnitude greater than fluxes reported in a natural upland (Table 4, McLain et al. 2008), a floodplain ecosystem in southeastern Arizona (Table 4, Harms and Grimm 2012) and elsewhere (Table 4, Billings et al. 2002; Bijoor et al. 2008; Groffman et al. 2009, Table 4) under similar soil wetting conditions. In other arid urban studies such as Hall et al. (2008) and Townsend-Small et al. (2011), N_2O fluxes from the finer textured washes were between 10 and 50 % of those observed at the sandy loam and loam washes in this study (Table 4). Finally, CH_4 fluxes were surprisingly larger and more variable than those reported in other arid and semi-arid studies (Table 4, Kaye et al. 2004; McLain and Martens 2006) but were similar to fluxes in humid urban and non-urban environments (Table 4, Groffman et al. 2009; Groffman and Pouyat 2009).

Substrate characteristics and physical controls on trace gas fluxes

The primary control on trace gas flux in these sites was moisture availability, and secondarily how the artificial wetting interacted with wash substrate. Sites with coarser substrates had lower water filled pore space, lower water holding capacity (Saxton et al. 1986), and smaller trace gas responses to wetting. Of secondary importance across sites was soil carbon content. Previous work in non-urban arid and semi-arid environments show that soil organic matter content (Sponseller 2007), rainfall amount (Sponseller 2007; Cable et al. 2008; McLain et al. 2008), antecedent moisture conditions and soil texture (Cable et al. 2008) control CO_2 fluxes. While all of these factors emerged as CO_2 flux controls in the ephemeral urban waterways of this study, here we show that soil texture largely controls the extent of priming and subsequent magnitude of soil respiration, presumably due to the close association between soil water holding content and soil organic matter (Hudson 1994; Saxton and Rawls 2006). Additional factors such as the quality and photodegradation of litter and soil organic matter (Brandt et al. 2009; Austin and Ballare 2010; Rutledge et al. 2010), might be important controls for C cycling in urban arid and semi-arid streams given the high solar radiation influx in the region (Unland et al. 1996; NREL 2008), and warrant further study.

N_2O fluxes were similarly controlled first by water availability, and secondarily by the rate of nitrogen cycling, net mineralization and nitrification rates and stocks of soil C and N, consistent with the "leaky pipe" model (Firestone et al. 1989). The higher variability in N_2O fluxes, mineralization and nitrification rates in the finer textured washes, coupled with simultaneous production of CO_2 and CH_4 gases indicate that as particle size decreases, the likelihood of variable redox states within the soil matrix increases. Simultaneous increases of CO_2 and N_2O fluxes in both instantaneous and post wetting responses, may not seem in harmony with our conceptual understanding as aerobic respiration, the process responsible for CO_2 production, depends on

Table 4 Trace gas losses from natural and urban arid and semi-arid ecosystems

Study	Location	Climate and biome (Koppen)	Habitat/soil type	Mean flux (min, max mass m^{-2} h^{-1})		
				mg CO_2–C	µg N_2O–N	µg CH_4–C
This study	Tucson, AZ	Subtropical dry arid (Sonoran desert)	Sandy ephemeral streams[a]	41.8 (7.4, 126)	5.3 (0, 45.4)	7.5 (−44.1, 56.1)
			Sandy loam ephemeral stream[a]	193.3 (2.4, 613.2)	243.9 (−0.4, 3,122)	−1.1 (−95.2, 37)
			Loam ephemeral stream[a]	329.5 (15.2–1,588.5)	235.6 (−250.6, 2540)	40.4 (−11.5, 266.2)
Allaire et al. (2008)	Quebec city, Canada		Urban turf grass lawns	60 (−59, 655)		
Bijoor et al. (2008)	Irvine, CA	Coastal mediterranean	Urban fescue and crabgrass lawn		(0, 21[b])	
			Urban fescue and crab grass lawn—artificially warmed		(0, 83.3)	
Billings et al. (2002)	Mercury, NV	Mid-latitude dry arid (Mohave desert)	Desert soils—under plants		(1.0, 1.8)	
			Desert soils—interspaces		0.7	
Bremer (2006)	Rocky Ford, KS		Perennial ryegrass turf grass fertilized		(−22, 407)	
Galbally et al. (2008)	Victoria, AU	Semiarid	Native Scrub		2.16	13.7
Groffman et al. (2009)	White mountain national forest, NH		Forest soil	(−10, 160)	(−80, 60)	(−167, 33)
Groffman and Pouyat (2009)	Baltimore LTER, MD		Urban grass			−7.3 (−31.9, 63.7)
			Urban and non-urban forest			−50.3 (−191.2, 286.8)
Hall et al. (2008)	Phoenix, AZ	Subtropical dry arid (desert)	Urban xeric landscapes		(<30)	
			Undeveloped desert		<20	
			Urbanized lawns		(18, 80)	
Harms and Grimm (2012)		Subtropical dry arid (Sonoran desert)	Floodplain soils		0.7 (<95)	
Kaye et al. (2004)	Fort collins, CO	Mid-latitude dry semiarid (steppe)	Urban lawns		27.4	(−17.1)
			Corn fields		<10 (<350)	(>−20)
			Native grass and wheat		(<4, 19)	−0.30 (−25, −55)
McCrackin et al. (2008)	Maricopa, AZ	Subtropical dry arid (desert)	Remnant non-developed urban soils—under plant	(∼8, 462)		
			Remnant non-developed urban soils interspaces	(∼8, 173)		

Table 4 continued

Study	Location	Climate and biome (Koppen)	Habitat/soil type	Mean flux (min, max mass m^{-2} h^{-1})		
				mg CO_2–C	µg N_2O–N	µg CH_4–C
McLain and Martens (2006)	San pedro river, AZ	Subtropical dry arid (Sonoran desert)	Soils under mesquite	(12, 121)	~(0, 15)	(−28, 0)
			Soils under sacaton	(7, 118)	~(0, 5)	(−27, >65)
			Soil between plants	(6, 113)	~(0, 5)	(−38, 0)
McLain et al. (2008)	Santa rita experimental range, AZ	Subtropical dry arid (Sonoran desert)		(<6, >116)	(<−6, >37)	(−31, >76)
Raciti et al. (2011)	Baltimore, MD, BES		Suburban lawn, not fertilized	23.2	−0.07	
			Suburban lawn, fertilized	43.4	2.0	
Sponseller (2007)	Sycamore creek, phoenix, AZ	Subtropical dry arid (desert)	Ephemeral stream—soil beneath plants	(50, 2,200)		
			Ephemeral stream—plant interspaces	(30, 690)		
Townsend-Small et al. (2011)	Pomona and irvine, CA	Coastal Mediterranean	Row crops		13.7	
			Corn fields		25.1	
			Urban turf lawn		26.3	
			Urban athletic fields		20.5	
Townsend-Small and Czimczik (2010)	Irvine, CA	Coastal Mediterranean	Park turf		92.9 (0, 720)	

[a] Mean includes pre-wetting fluxes

the presence of oxygen (O_2) to act as a terminal electron acceptor whereas N_2O production via denitrification is an anaerobic process. As McLain et al. (2008) suggest, high rates of respiration and or gas diffusional limitations may create localized anaerobic microsites where denitrification can occur (Parkin 1987). Production of N_2O may also be derived from organic N nitrification and subsequent nitrifier denitrification, an aerobic process, and thus not be dependent on the formation of anaerobic microsites (McLain and Martens 2005; Kool et al. 2011). In addition, N_2O production may be derived from activity of heterotrophic fungi, which are more adapted to lower soil moisture than bacteria and have been shown to generate substantial fluxes in this region (Jensen and Hauggaard-Nielsen 2003; Pietikåinen et al. 2005; McLain and Martens 2006). Finally, recent studies suggest that DNRA is more ubiquitous than previously thought and may also produce N_2O gases as nitrate is being reduced to ammonium (Baggs 2011).

We observed enhanced CH_4 uptake in the sand and sandy loam washes suggesting methanotrophic conditions in these coarser, drier and more C and N limited soils; while in the loam washes, we observed enhanced CH_4 production (Figs. 2h, 3c) suggesting methanogenic conditions in these finer textured, wetter and less nutrient limited soils (Angel 2010; Kim et al. 2012). Despite these patterns, the high variability in CH_4 fluxes over time (Fig. 2h), particularly in the loam washes, indicate the possibility of concurrent methanotrophy and methanogenesis. While the co-occurrence of these processes may seem contradictory, studies point to chemical pathways for oxic CH_4 production in soils (Hurkuck et al. 2012; Jugold et al. 2012) as well as aerobic CH_4 production in plant and animal tissues (Keppler et al. 2009). Although exact chemical pathways for oxic CH_4 production in soils remain to been identified, Hurkuck et al. (2012) and Jugold et al.(2012) demonstrate that abiotic oxic CH_4 production varies with UV radiation, temperature, soil organic matter quality and quantity, and soil moisture regime. A study by McLain et al. (2008) in a semi-arid rangeland ecosystem near Tucson, AZ observed similarly high CH_4 production as we document here, which was linked to the activity of methanogens residing in the guts of soil invertebrates and termites, a biotic process that may, in part, explain the patterns we observed. In addition, Angel (2010) documents the co-existence of methanogenic and methanotrophic bacteria within oxic biological soil crusts of Israeli desert soils. While the activity of methanogens was reduced under aerobic conditions, the rapid development of anoxic micro sites following soil wetting allowed for a significant, albeit somewhat depressed flux of CH_4 during prevailing oxic conditions. It is plausible that the rapid development of anoxic micro-sites, coupled with an enhanced availability of labile carbon following wetting (Kim et al. 2012) may allow for the co-occurrence of methanogenesis and methanotrophy in the loam washes, and might enhance CH_4 production between rainfall events.

Implications of ephemeral stream channel biogeochemistry to management

Our data indicates that ephemeral urban channels provide ecosystem services beyond those of a flow through stormwater management system; they are hot spots where disproportionately high rates of biogeochemical cycling occur during hot moments that have potentially large effects on basin scale N balances. For example, in their study of the Central Arizona Phoenix (CAP) LTER, Baker et al. (2001) found that more than 50 % of the N loss from arid urban ecosystems occurs via denitrification. Preliminary estimates for catchment wide N-budgets in the Tucson basin made using findings from this study and coupled with our previous work (Gallo et al. 2012a, b, 2013) indicate that as much as 500 mg m^{-2} of N may be lost via N_2O gas flux from ephemeral streams to the atmosphere in response to summertime rainfall events of 15 mm in depth or less. Thus, these small and often non-runoff producing events can jump start biogeochemical processes by releasing soil moisture limitations, particularly in sites like the sandy loam washes which are primarily water limited.

The temporal distribution of biogoeochemical hot moments in ephemeral waterways remains to be quantified over an entire season. Our previous work indicates that following a wetting event, soil moisture quickly resets to pre-wetting conditions (Gallo et al. 2012a, b); and here we show that trace gas fluxes vary primarily with soil moisture, and that antecedent moisture conditions result in variable redox states and nutrient cycling pathways. Combined, our work suggests that mid and late summer trace gas fluxes in response to wetting might approximate early season trace gas flux rates as soil moisture decreases and the intervening period between rainfall events increases.

Because urban runoff is increasingly managed to augment limited water supplies in arid and semi-arid regions, reducing delivery of nitrate areas of focused recharge is critical to sustaining groundwater resources (Carlson et al. 2011). Our previous work suggest that larger rainfall events produce runoff for potential recharge and transport N out of urban catchments (Gallo et al. 2012a, 2013), while this study indicates that small rainfall events, like that simulated here, have the potential to remove N from ephemeral channels that serve as primary recharge zones (Pool 2005; Blasch et al. 2010). Through removal of N via trace gas losses, these urban biogeochemical hotspots, which include pervious streams and green storm water management infrastructure, may improve downstream water quality by reducing the delivery of NO_3 to areas of groundwater recharge. However, tradeoffs related to greenhouse gas production warrant further evaluation.

Conclusions

In this study we show that trace gas losses in response to a rainfall event are largely controlled by the textural characteristics of ephemeral stream substrate which alter how soil water, carbon and nitrogen interact and flux through the soil system between wetting events. We observed shifts in trace gas flux controls, from soil moisture, soil C and soil N limitations on trace gas production in coarser stream soils, to biologically favorable antecedent conditions due to higher soil C and N availability, soil organic matter and subsequent soil moisture retention in the finer textured stream soils. The patterns in trace gas fluxes over the soil texture gradient suggest that redox states within the stream channel substrate become more variable as particle size decreases. Surprisingly, here we document extremely high fluxes of CO_2 and N_2O in response to stream channel wetting, indicating that ephemeral urban streams have the potential to process large loads of C and N following non-runoff producing rainfall pulses. The fluxes we document are elevated in comparison to undeveloped deserts, urbanized uplands and more humid systems. Collectively, our study suggests that the urban ephemeral stream channels examined here comprise urban biogeochemical hot spots that have short, pronounced hot moments when a rainfall pulse follows an extensive period of drought; and that urban streams have the potential to significantly alter C and N fluxes and pathways of urban environments.

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References

Allaire SE, Dufour-L'Arrivée C, Lafond JA, Lalancette R, Brodeur J (2008) Carbon dioxide emissions by urban turfgrass areas. Can J Soil Sci 88:529–532. doi:10.4141/CJSS07043

Angel R (2010) Methane turnover in desert soils. Department of biology, Max-Planck Institute for Terrestrial Microbiology, Philipps-Universität Marburg, Marburg, p 191

Austin AT, Ballare CL (2010) Dual role of lignin in plant litter decomposition in terrestrial ecosystems. Proc Natl Acad Sci 107:4618–4622. doi:10.1073/pnas.0909396107

Austin AT, Yahdjian L, Stark JM, Belnap J, Porporato A, Norton U, Ravetta DA, Schaeffer SM (2004) Water pulses and biogeochemical cycles in arid and semiarid ecosystems. Oecologia 141:221–235. doi:10.1007/s00442-004-1519-1

Baggs EM (2011) Soil microbial sources of nitrous oxide: recent advances in knowledge, emerging challenges and future direction. Curr Opin Environ Sustain 3:321–327. doi:10.1016/j.cosust.2011.08.011

Baker LA, Hope D, Xu Y, Edmonds J, Lauver L (2001) Nitrogen balance for the central arizona–phoenix (CAP) ecosystem. Ecosystems 4:582–602. doi:10.1007/s10021-001-0031-2

Belnap J, Welter JR, Grimm NB, Barger N, Ludwig JA (2005) Linkages between microbial and hydrologic processes in arid and semiarid watersheds. Ecology 86:298–307. doi:10.1890/03-0567

Bijoor NS, Czimczik CI, Pataki DE, Billings SA (2008) Effects of temperature and fertilization on nitrogen cycling and community composition of an urban lawn. Glob Change Biol 14:2119–2131. doi:10.1111/j.1365-2486.2008.01617.x

Billings SA, Schaeffer SM, Evans RD (2002) Trace N gas losses and N mineralization in Mojave desert soils exposed to elevated CO_2. Soil Biol Biochem 34:1777–1784. doi:10.1016/S0038-0717(02)00166-9

Birch HF (1958) The effect of soil drying on humus decomposition and nitrogen availability. Plant Soil 10:9–31. doi:10.1007/BF01343734

Blasch K, Ferré T, Vrugt J (2010) Environmental controls on drainage behavior of an ephemeral stream. Stoch Envirom Res Risk Assess 24:1077–1087. doi:10.1007/s00477-010-0398-8

Bowling DR, Grote EE, Belnap J (2011) Rain pulse response of soil CO_2 exchange by biological soil crusts and grasslands of the semiarid Colorado Plateau, United States. J Geophys Res Biogeosci 116:G03028. doi:10.1029/2011JG001643

Brandt LA, Bohnet C, King JY (2009) Photochemically induced carbon dioxide production as a mechanism for carbon loss from plant litter in arid ecosystems. J Geophys Res Biogeosci 114:G02004. doi:10.1029/2008JG000772

Bremer DJ (2006) Nitrous oxide fluxes in turfgrass. J Environ Qual 35:1678–1685. doi:10.2134/jeq2005.0387

Cable JM, Huxman TE (2004) Precipitation pulse size effects on Sonoran desert soil microbial crusts. Oecologia 141:317–324. doi:10.1007/s00442-003-1461-7

Cable J, Ogle K, Williams D, Weltzin J, Huxman T (2008) Soil texture drives responses of soil respiration to precipitation pulses in the Sonoran desert: implications for climate change. Ecosystems 11:961–979. doi:10.1007/s10021-008-9172-x

Carlson MA, Lohse KA, McIntosh JC, McLain JET (2011) Impacts of urbanization on groundwater quality and recharge in a semi-arid alluvial basin. J Hydrol 409:196–211. doi:10.1016/j.jhydrol.2011.08.020

Conant RT, Dalla-Betta P, Klopatek CC, Klopatek JM (2004) Controls on soil respiration in semiarid soils. Soil Biol Biochem 36:945–951. doi:10.1016/j.soilbio.2004.02.013

DeCoster J (1998) Overview of factor analysis. Retrieved June 8, 2009 from http://www.stat-help.com/notes.html

Ezcurra E (2006) Global deserts outlook. UNEP, Division of Early Warning and Assessment, Nairobi

Fenn ME, Baron JS, Allen EB, Rueth HM, Nydick KR, Geiser L, Bowman WD, Sickman JO, Meixner T, Johnson DW (2003) Ecological effects of nitrogen deposition in the western United States. Bioscience 53:404–420. doi:10.1641/0006-3568(2003)053[0404%3AEEONDI]2.0.CO%3B2

Firestone M, Davidson E, Andreae M, Schimel D (1989) Microbiological basis of NO and N_2O production and consumption in soil. In: Andreae M, Schimel D (eds) Exchange of trace gases between terrestrial ecosystems and the atmosphere. Wiley, Chichester, pp 7–21

Galbally IE, Kirstine WV, Meyer CP, Wang YP (2008) Soil-atmosphere trace gas exchange in semiarid and arid zones. J Environ Qual 37:599–607. doi:10.2134/jeq2006.0445

Gallo EL (2011) Patterns and controls of urban runoff quantity and quality in catchments of the semi-arid southwest. Department of Hydrology and Water Resources, University of Arizona, Tucson, p 240

Gallo EL, Brooks P, Lohse KA, McLain JE (2012a) Temporal patterns and controls on runoff magnitude and solution chemistry of urban catchments in the semi-arid southwest. Hydrol Process. doi:10.1002/hyp.9199

Gallo EL, Lohse KA, Brooks P, McLain JE (2012b) Quantifying the effects of stream channels on storm water quality in a semi-arid urban environment. J Hydrol. doi:10.1016/j.jhydrol.2012.08.047

Gallo EL, Brooks PD, Lohse KA, McLain JET (2013) Land cover controls on summer discharge and runoff solution chemistry of semi-arid urban catchments. J Hydrol 485:37–53. doi:10.1016/j.jhydrol.2012.11.054

Gee GW, Bauder JW (1986) Particle-size analysis. In: Klute A (ed) Methods of soil analysis. Part 1 - physical and mineralogical properties. Soil Science Society of America Book Series 5, Madison, WI, pp 383–411

Goddard MA, Dougill AJ, Benton TG (2010) Scaling up from gardens: biodiversity conservation in urban environments. Trends Ecol Evol 25:90–98. doi:10.1016/j.tree.2009.07.016

Grimm NB, Faeth SH, Golubiewski NE, Redman CL, Wu JG, Bai XM, Briggs JM (2008a) Global change and the ecology of cities. Science 319:756–760. doi:10.1126/science.1150195

Grimm NB, Foster D, Groffman P, Grove JM, Hopkinson CS, Nadelhoffer KJ, Pataki DE, Peters DPC (2008b) The changing landscape: ecosystem responses to urbanization and pollution across climatic and societal gradients. Front Ecol Environ 6:264–272. doi:10.1890/070147

Groffman PM, Pouyat RV (2009) Methane uptake in urban forests and lawns. Environ Sci Technol 43:5229–5235. doi:10.1021/es803720h

Groffman P, Hardy J, Fisk M, Fahey T, Driscoll C (2009) Climate variation and soil carbon and nitrogen cycling processes in a northern hardwood forest. Ecosystems 12:927–943. doi:10.1007/s10021-009-9268-y

Grossman RB, Reinsch TG (2002) Bulk density and linear extensibility. In: Dane JH, Topp GC, Campbell GS (eds) Methods of soil analysis. Part 4 - physical methods. Soil Science Society of America, Madison, WI, pp 201–228

Guido Z (2008) Understanding the southwestern monsoon. Southwest Climate Change Network

Hall SJ, Huber D, Grimm NB (2008) Soil N_2O and NO emissions from an arid, urban ecosystem. J Geophys Res Biogeosci 113:G01016. doi:10.1029/2007JG000523

Hall SJ, Ahmed B, Ortiz P, Davies R, Sponseller RA, Grimm NB (2009) Urbanization alters soil microbial functioning in the Sonoran desert. Ecosystems 12:654–671. doi:10.1007/s10021-009-9249-1

Harms TK, Grimm NB (2008) Hot spots and hot moments of carbon and nitrogen dynamics in a semiarid riparian zone. J Geophys Res Biogeosci 113:1–14. doi:10.1029/2007JG000588

Harms TK, Grimm NB (2012) Responses of trace gases to hydrologic pulses in desert floodplains. J Geophys Res Biogeosci 2005–2012:117. doi:10.1029/2011JG001775

Hart SC, Stark JM, Davidson EA, Firestone MK (1994) Nitrogen mineralization, immobilization, and nitrification. In: Weaver R, Angle S, Bottomley P, Bedzicek D, Smith S, Tabatabai A, Wollum A, Mickelson SH, Bigham JM (eds) Methods of soil analysis, part 2—microbiological and biochemical properties. Soil Science Society of America, Madison, pp 985–1018

Houser P, Goodrich D, Syed K, U.S.D.A. ARS (2000) Runoff, precipitation and soil moisture at walnut gulch. In: Grayson R, Blöschl GN (eds) Spatial patterns in catchment hydrology: observations and modelling. Cambridge University Press, Cambridge, pp 125–157

Hudson BD (1994) Soil organic matter and available water capacity. J Soil Water Conserv 49:189–194

Hurkuck M, Althoff F, Jungkunst HF, Jugold A, Keppler F (2012) Release of methane from aerobic soil: an indication of a novel chemical natural process? Chemosphere 86:684–689. doi:10.1016/j.chemosphere.2011.11.024

Huxman TE, Snyder KA, Tissue D, Leffler AJ, Ogle K, Pockman WT, Sandquist DR, Potts DL, Schwinning S (2004) Precipitation pulses and carbon fluxes in semiarid and arid ecosystems. Oecologia 141:254–268. doi:10.1007/s00442-004-1682-4

Jarvis P, Rey A, Petsikos C, Wingate L, Rayment M, Pereira J, Banza J, David J, Miglietta F, Borghetti M, Manca G, Valentini R (2007) Drying and wetting of Mediterranean soils stimulates decomposition and carbon dioxide emission: the "Birch effect". Tree Physiol 27:929–940. doi:10.1093/treephys/27.7.929

Jensen ES, Hauggaard-Nielsen H (2003) How can increased use of biological N_2 fixation in agriculture benefit the environment? Plant Soil 252:177–186. doi:10.1023/A:1024189029226

Jugold A, Althoff F, Hurkuck M, Greule M, Lelieveld J, Keppler F (2012) Non-microbial methane formation in oxic soils. Biogeosci Discuss 9:11961–11987. doi:10.5194/bgd-9-11961-2012

Kaiser HF (1958) The varimax criterion for analytic rotation in factor-analysis. Psychometrika 23:187–200. doi:10.1007/BF02289233

Kaye JP, Burke IC, Mosier AR, Pablo Guerschman J (2004) Methane and nitrous oxide fluxes from urban soils to the atmosphere. Ecol Appl 14:975–981. doi:10.1890/03-5115

Kaye JP, Groffman PM, Grimm NB, Baker LA, Pouyat RV (2006) A distinct urban biogeochemistry? Trends Ecol Evol 21:192–199. doi:10.1016/j.tree.2005.12.006

Kennedy J (2007) Changes in storm runoff with urbanization: the role of pervious areas in a semi-arid environment. Department of Hydrology and Water Resources, The University of Arizona, Tucson, p 111

Keppler F, Boros M, Frankenberg C, Lelieveld J, McLeod A, Pirttilä AM, Röckmann T, Schnitzler J-P (2009) Methane formation in aerobic environments. Environ Chem 6:459–465. doi:10.1071/EN09137

Kim DG, Vargas R, Bond-Lamberty B, Turetsky MR (2012) Effects of soil rewetting and thawing on soil gas fluxes: a review of current literature and suggestions for future research. Biogeosciences 9:2459–2483. doi:10.5194/bg-9-2459-2012

Kool DM, Dolfing J, Wrage N, Van Groenigen JW (2011) Nitrifier denitrification as a distinct and significant source of nitrous oxide from soil. Soil Biol Biochem 43:174–178. doi:10.1016/j.soilbio.2010.09.030

Larson EK (2010) Water and nitrogen in designed ecosystems: biogeochemical and economic consequences. Biology, Arizona State University, Tempe, p 275

Lehman A, O'Rouke N, Hatcher L, Stepanski EJ (2005) JMP for basic univariate and multivariate statistics a step-by-step guide. SAS Press, Cary

Levick LR, Goodrich DC, Hernandez M, Fonseca J, Semmens DJ, Stromberg JC, Tluczek M, Leidy RA, Scianni M, Guertin DP (2008) The ecological and hydrological significance of ephemeral and intermittent streams in the arid and semi-arid American southwest. US Environmental Protection Agency, Office of Research and Development, Washington, DC

Lewis DB, Grimm NB (2007) Hierarchical regulation of nitrogen export from urban catchments: interactions of storms and landscapes. Ecol Appl 17:2347–2364. doi:10.1890/06-0031.1

Lohse KA, Hope D, Sponseller R, Allen JO, Grimm NB (2008) Atmospheric deposition of carbon and nutrients across an and metropolitan area. Sci Total Environ 402:95–105. doi:10.1016/j.scitotenv.2008.04.044

Loik ME, Breshears DD, Lauenroth WK, Belnap J (2004) A multi-scale perspective of water pulses in dryland ecosystems: climatology and ecohydrology of the western USA. Oecologia 141:269–281. doi:10.1007/s00442-004-1570-y

Lorenz K, Lal R (2009) Biogeochemical C and N cycles in urban soils. Environ Int 35:1–8. doi:10.1016/j.envint.2008.05.006

Marañón-Jiménez S, Castro J, Kowalski A, Serrano-Ortiz P, Reverter B, Sánchez-Cañete E, Zamora R (2011) Post-fire soil respiration in relation to burnt wood management in a Mediterranean mountain ecosystem. For Ecol Manage 261:1436–1447. doi:10.1016/j.foreco.2011.01.030

McClain ME, Boyer EW, Dent CL, Gergel SE, Grimm NB, Groffman PM, Hart SC, Judson WH, Johnston CA, Mayorga E, McDowell WH, Pinay G (2003) Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6:301–312. doi:10.1007/s10021-003-0161-9

McCrackin ML, Harms TK, Grimm NB, Hall SJ, Kaye JP (2008) Responses of soil microorganisms to resource availability in urban, desert soils. Biogeochemistry 87:143–155. doi:10.1007/s10533-007-9173-4

McLain JE, Martens DA (2005) Nitrous oxide flux from soil amino acid mineralization. Soil Biol Biochem 37:289–299. doi:10.1016/j.soilbio.2004.03.013

McLain JET, Martens DA (2006) Moisture controls on trace gas fluxes in semiarid riparian soils. Anglais 70:367–377. doi:10.2136/sssaj 2005.0105

McLain JET, Kepler TB, Ahmann DM (2002) Belowground factors mediating changes in methane consumption in a forest soil under elevated CO_2. Glob Biogeochem Cycles 16:1050. doi:10.1029/2001GB001439

McLain JET, Martens DA, McClaran MP (2008) Soil cycling of trace gases in response to mesquite management in a semiarid grassland. J Arid Environ 72:1654–1665. doi:10.1016/j.jaridenv.2008.03.003

Mendez A, Goodrich DC, Osborn HB (2003) Rainfall point intensities in an air mass thunderstorm environment: walnut gulch, Arizona. JAWRA 39:611–621. doi:10.1111/j.1752-1688.2003.tb03679.x

Navarro-García F, Casermeiro MÁ, Schimel JP (2012) When structure means conservation: effect of aggregate structure in controlling microbial responses to rewetting events. Soil Biol Biochem 44:1–8. doi:10.1016/j.soilbio.2011.09.019

NREL (2008) United States concentrating solar power—direct normal. National Renewable Energy Laboratory, U. S. Department of Energy, Washington, DC

Parkin TB (1987) Soil microsites as a source of denitrification variability. Anglais 51:1194–1199. doi:10.2136/sssaj1987.03615995005100050019x

Pataki DE, Carreiro MM, Cherrier J, Grulke NE, Jennings V, Pincetl S, Pouyat RV, Whitlow TH, Zipperer WC (2011) Coupling biogeochemical cycles in urban environments: ecosystem services, green solutions, and misconceptions. Front Ecol Environ 9:27–36. doi:10.1890/090220

Pietikåinen J, Pettersson M, Bååth E (2005) Comparison of temperature effects on soil respiration and bacterial and fungal growth rates. FEMS Microbiol Ecol 52:49–58. doi:10.1016/j.femsec.2004.10.002

Pool DR (2005) Variations in climate and ephemeral channel recharge in southeastern Arizona, United States. Water Resour Res 41:W11403. doi:10.1029/2004WR003255

Pouyat R, Groffman P, Yesilonis I, Hernandez L (2002) Soil carbon pools and fluxes in urban ecosystems. Environ Pollut 116(Supplement 1):S107–S118. doi:10.1016/S0269-7491(01)00263-9

Raciti SM, Burgin AJ, Groffman PM, Lewis DN, Fahey TJ (2011) Denitrification in suburban lawn soils. J Environ Qual 40:1932–1940. doi:10.2134/jeq 2011.0107

Reynolds J, Kemp P, Ogle K, Fernández R (2004) Modifying the 'pulse–reserve' paradigm for deserts of North America: precipitation pulses, soil water, and plant responses. Oecologia 141:194–210. doi:10.1007/s00442-004-1524-4

Roach WJ, Grimm NB (2011) Denitrification mitigates N flux through the stream-floodplain complex of a desert city. Ecol Appl 21:2618–2636. doi:10.1890/10-1613.1

Rutledge S, Campbell DI, Baldocchi D, Schipper LA (2010) Photodegradation leads to increased carbon dioxide losses from terrestrial organic matter. Glob Change Biol 16:3065–3074. doi:10.1111/j.1365-2486.2009.02149.x

Sarrantonio M, Doran JW, Liebig MA, Halvorson JJ (1996)On-farm assessment of soil quality and health. In: Doran JW, Jones AJ (eds) Methods for assessing soil quality. SSSA Spec. Publ. 49. SSSA, Madison, WI, pp 83–106

Saxton KE, Rawls WJ (2006) Soil water characteristic estimates by texture and organic matter for hydrologic solutions. Soil Sci Soc Am J 70:1569–1578. doi:10.2136/sssaj2005.0117

Saxton KE, Rawls WJ, Romberger JS, Papendick RI (1986) Estimating generalized soil-water characteristics from texture. Soil Sci Soc Am J 50:1031–1036. doi:10.2136/sssaj1986.03615995005000040039x

Schwinning S, Sala O (2004) Hierarchy of responses to resource pulses in arid and semi-arid ecosystems. Oecologia 141:211–220. doi:10.1007/s00442-004-1520-8

Šimůnek J, van Genuchten MT, Šejna M (2005) The Hydrus-1D software package for simulating the one-dimensional movement of water, heat, and multiple solutes in variably-saturated media, version 3.0, HYDRUS Software Series, 1st edn. Department of Environmental Sciences, University of California Riverside, Riverside

Snyder KA, Williams DG (2000) Water sources used by riparian trees varies among stream types on the San Pedro River, Arizona. Agric For Meteorol 105:227–240. doi:10.1016/S0168-1923(00)00193-3

Sponseller RA (2007) Precipitation pulses and soil CO_2 flux in a Sonoran desert ecosystem. Glob Change Biol 13:426–436. doi:10.1111/j.1365-2486.2006.01307.x

Sponseller RA, Fisher SG (2008) The influence of drainage networks on patterns of soil respiration in a desert catchment. Ecology 89:1089–1100. doi:10.1890/06-1933.1

Theobald DM, Travis WR, Drummond MA, Gordon ES (2013) The changing southwest. In: Garfin G, Jardine A, Merideth R, Black M, LeRoy S (eds) Assessment of climate change in the southwest United States: a report prepared for the national climate assessment, a report by the southwest climate alliance. Island Press, Washington, DC, pp 37–55

Townsend-Small A, Czimczik CI (2010) Carbon sequestration and greenhouse gas emissions in urban turf. Geophys Res Letters 37:L02707. doi:10.1029/2009GL041675

Townsend-Small A, Pataki DE, Czimczik CI, Tyler SC (2011) Nitrous oxide emissions and isotopic composition in urban and agricultural systems in southern California. J Geophys Res Biogeosci 116:G01013. doi:10.1029/2010JG001494

Unger S, Máguas C, Pereira JS, David TS, Werner C (2010) The influence of precipitation pulses on soil respiration—assessing the "Birch effect" by stable carbon isotopes. Soil Biol Biochem 42:1800–1810. doi:10.1016/j.soilbio.2010.06.019

Unland HE, Houser PR, Shuttleworth WJ, Yang Z-L (1996) Surface flux measurement and modeling at a semi-arid Sonoran desert site. Agric For Meteorol 82:119–153. doi:10.1016/0168-1923(96)02330-1

Zar J (1999) Biostatistical analysis. Prentice Hall, New Jersey

Zhang Y, Wei H, Nearing MA (2011) Effects of antecedent soil moisture on runoff modeling in small semiarid watersheds of southeastern Arizona. Hydrol Earth Syst Sci Discuss 8:6227–6256. doi:10.5194/hessd-8-6227-2011

Zhu W-X, Dillard N, Grimm N (2005) Urban nitrogen biogeochemistry: status and processes in green retention basins. Biogeochemistry 71:177–196. doi:10.1007/s10533-005-0683-7