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Nutrient Cycling in Grassed Roadside Ditches and Lawns in a Suburban Watershed

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Abstract

Roadside ditches are ubiquitous in developed landscapes. They are designed to route water from roads for safety, with little consideration of water quality or biogeochemical implications in ditch design and minimal data on environmental impacts. We hypothesize that periodic saturation and nutrient influxes may make roadside ditches hotspots for nitrogen (N) removal via denitrification as well as biological production of the greenhouse gases (GHGs) nitrous oxide (N_2O), methane (CH_4), and carbon dioxide (CO_2). Research sites included 12 grassed ditches and adjacent lawns with varying fertilization in a suburban watershed in central New York, where lawns represented a reference with similar soils as ditches but differing hydrology. We measured potential denitrification using the denitrification enzyme assay in fall 2014 and GHG fluxes using in situ static chambers between summer 2014 and 2015, including sample events after storms. Potential denitrification in ditches was significantly higher than in lawns, and rates were comparable to those in stream riparian areas, features traditionally viewed as denitrification hotspots. Ditches had higher rates of CH_4 emissions, particularly sites that were wettest. Lawns were hotspots for N_2O and CO_2 respiratory emissions, which were driven by nutrient availability and fertilizer application. Extrapolating up to the watershed, ditches have the potential to remove substantial N loads via denitrification if managed optimally. Ditch GHG emissions extrapolated across the watershed were minimal given their much smaller area compared with lawns, which were the greater contributor of GHGs. These findings suggest that roadside ditches may offer new management opportunities for mitigating nonpoint source N pollution in residential watersheds.

Core Ideas

- Greenhouse gases and denitrification were studied in grassed road ditches and lawns.
- Nitrous oxide emissions were higher in lawns, particularly in fertilized lawns.
- Ditches were hotspots for CH_4 emissions and potential denitrification.
- Ditches could be better managed to promote beneficial N removal and minimize GHGs.

Suburban and exurban areas are a growing fraction of the landscape in the United States, increasing in area by more than 500% since 1950 (Brown et al., 2005). With increased development has come construction of roads for transportation, with public roads in the United States totaling 6.3 million km and covering approximately 1% of land by area (NRC, 1997). The majority of these roads are flanked by ditches, which help quickly route stormwater off of roads for safety purposes. This introduction of roadside ditches into watersheds causes substantial manipulation of landscape hydrology (Buchanan et al., 2013; Forman et al., 2003).

Even in watersheds with low development density, roadside ditches can intercept approximately 30% of the runoff generated in the watershed (Buchanan et al., 2013). This landscape modification results in substantial redistribution of water and associated solutes or particulates in the watershed. Previous research in rural watersheds in central New York has found that roadside ditches negatively influence downstream water quality, serving as "efficient conduits" of sediment, phosphorus, and *Escherichia coli* to receiving streams (Buchanan et al., 2013; Falbo et al., 2013). Although there has been some investigation of pollutant transport in ditches, there has been little work investigating biogeochemical transformations of various solutes in ditch soils. With their intermittent pulses of water and associated solutes, grassed ditches could be hotspots for biogeochemical processes that remediate pollutants and/or generate greenhouse gases (GHGs).

Nitrogen (N) is a nutrient of particular interest because of its potential to contribute to GHG production and downstream water quality impairment. In suburban watersheds, which often have a high density of roads and corresponding ditches or swales, potential N sources for stormwater include fertilizer applied to lawns (Raciti et al., 2011b) as well as atmospheric deposition of N from sources such as automobile exhaust (Bettez and Groffman, 2013). Opportunities for removing N from biologically available forms like nitrate (NO_3^-) include assimilation by vegetation or microbes as well as denitrification.

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Abbreviations: CO_2eq, CO_2 equivalents; DOC, dissolved organic carbon; GHG, greenhouse gas; GWP, global warming potential; OM, organic matter; VWC, volumetric water content.

The microbially mediated process of denitrification, which transforms NO_3^- into nitrous oxide (N_2O) or dinitrogen gas (N_2) (Tiedje, 1994), may be occurring in vegetated ditches and acting as an important NO_3^- removal pathway. However, N_2O is a GHG with 298 times the global warming potential (GWP) of carbon dioxide (CO_2) on a 100-yr timescale. In addition to being a possible by-product of denitrification, N_2O can also be produced by nitrification (Firestone and Davidson, 1989). Previous research has documented the occurrence of denitrification hotspots (i.e., zones of increased denitrification rates relative to surroundings) in the environment where there is the convergence of nutrient inputs and high soil wetness (Groffman et al., 2009a; McClain et al., 2003). Although roadside ditches are typically characterized by flashy, intermittent flow (Buchanan et al., 2013), there could be periods of time during and shortly after storm events that satisfy the conditions needed for denitrification. Some of these conditions (e.g., high soil moisture leading to the development of anoxia and the availability of carbon [C] and N substrate) also influence production of other GHGs. Methane (CH_4), which is produced by soil microbes when there is high soil moisture and ample soil C, is a GHG with 25 times the GWP of CO_2 (Walter and Heimann, 2000). Carbon dioxide is produced by multiple respiratory processes occurring in soils (Schlesinger and Andrews, 2000).

Denitrification and GHG production have been studied in various features of suburban landscapes. Previous research has focused on lawns (Byrne et al., 2008; Groffman and Pouyat, 2009; Kaye et al., 2004; Livesley et al., 2010), stormwater control measures such as detention basins (Bettez and Groffman, 2012; Grover et al., 2013; McPhillips and Walter, 2015), and streams and their riparian areas (Groffman and Crawford, 2003; Kaushal et al., 2008). Although roadside ditches are a common feature in suburban landscapes, there is limited knowledge of their water quality impacts and virtually no understanding of their role in nutrient cycling. Research in agricultural ditches has documented high N removal potential via denitrification (Roley et al., 2012), although these ditches are typically more continuously flowing compared with roadside ditches, and thus nutrient cycling dynamics may differ. An improved understanding of biogeochemical dynamics in roadside ditches can inform better ditch design and landscape management practices, given that ditches are a human-made feature.

The primary objective of this study was to quantify denitrification and GHG emissions in 12 grassed roadside ditches (hereafter "ditches") and adjacent lawns, given that lawns have similar soil properties and management to the ditches in this watershed but do not receive the stormflows that the ditches do. Additionally, we sought to link any observed patterns in biogeochemical processes to environmental factors (e.g., soil moisture), landscape characteristics, or management practices; to estimate the impact of ditches and lawns on denitrification and GHG processes across the watershed; and to recommend improvements to management of suburban landscapes to maximize water quality services like denitrification and minimize GHG emissions.

Materials and Methods

The experimental sites were located in the watershed of a first-order (unnamed) stream in Tompkins County, NY (within the town of Ithaca and villages of Cayuga Heights and Lansing; 42.477° N, −76.485° W) that drains to Cayuga Lake (Fig. 1). This region is characterized by a temperate climate, with an average annual temperature of 8.1°C (range, −9.2 to 26.6°C) and average annual precipitation of 947 mm (NRCC, 2016). The 293-ha, mixed-use watershed is representative of many suburban watersheds across the northeastern United States, containing 40% developed land (including residential, school, and commercial land), 44% forest, 8% wetland and open water, and 8% pasture. Existing stormwater infrastructure in the watershed includes detention basins and ponds, ditches, and minimal storm sewers, all of which discharge within the watershed (Easton and Petrovic, 2008). Most ditches in the watershed, particularly those adjacent to lawns on residential or commercial property, are vegetated with turfgrass and mowed at the same frequency as the lawns. Soils in the watershed are mapped as predominantly silt loam (USDA, 2015), although textural analysis found lawn and ditch soils to be predominantly sandy loam (Supplemental Table S1).

Potential experimental sites were identified through use of a homeowner survey. Information was mailed to homes within the

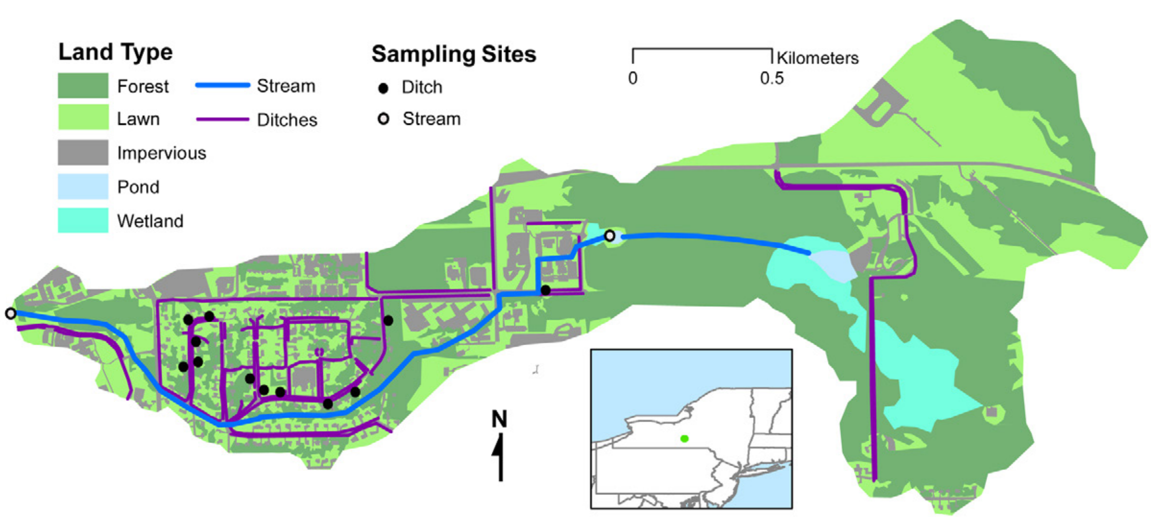

Fig. 1. Map of the study watershed displaying land cover types and locations of ditch and stream sampling sites. Inset map displays location of the watershed in New York State.

study watershed, and homeowners interested in allowing their ditch and adjacent lawn to be a research site were directed to an online survey form. The survey was granted an exemption from the Cornell Institutional Review Board. It solicited information on yard maintenance (including fertilizer application) as well as road ditch conditions, such as how often it contained flowing water. Using this information, 12 ditches and adjacent lawns (an example site photograph is shown in Fig. 2) were selected as experimental sites such that they represented a range of moisture regimes and lawn care practices (Supplemental Table S1).

Denitrification was measured using the denitrification enzyme assay, which assesses potential denitrification when ample C and NO_3^- are available (Groffman et al., 1999). Soil samples were taken at each ditch in October 2014. At each ditch, flux measurements were made at two locations within the ditch and at two locations in the lawn adjacent to the ditch. For each of the two within-ditch and two lawn locations, five soil cores (5 cm long, 2 cm diameter) were collected and homogenized; from this soil, two replicate subsamples of 5 to 6 g were used for the assay. Assays were conducted in 125-mL glass serum bottles following the methodology of McPhillips and Walter (2015). Briefly, nutrient media containing NO_3^- and glucose was added, along with acetylene to block conversion of N_2O to N_2 gas. Gas samples for N_2O analysis were taken at 20-min intervals over 60 min and analyzed using an Agilent 6890N gas chromatograph (Agilent Technologies Inc.) equipped with a HP 7694 Headspace Autosampler (Hewlett-Packard Co.). Nitrous oxide separation was performed using a Supel-Q PLOT capillary column (Supelco Inc.) and an electron capture detector. Calibration curves were made using a series of standard gas mixes (Airgas Inc.). Gas fluxes were calculated by determining the linear slope of the concentrations of the four time points (Rochette and Bertrand, 2008). Nitrous oxide fluxes were converted to denitrification rates in mg N kg soil^{-1} h^{-1} using the dry mass of the soil subsamples.

Fluxes of CO_2, CH_4, and N_2O were measured in the ditches on six dates between July 2014 and June 2015, which were chosen such that they spanned a range of temperature and moisture conditions (Supplemental Table S2). Several GHG sampling events occurred the day after a storm event during which ditches were flowing. Three-day antecedent precipitation for the sampling dates ranged from 0.28 to 3.81 cm, and soil temperature at time of sampling ranged from 7.4 to 26.8°C. On each sampling event (Supplemental Table S2), GHG fluxes were measured over two consecutive days in which there was minimal average temperature change and no new precipitation. Six ditch sites were measured on each of the 2 d during the morning.

Fluxes were measured using in situ static white PVC chambers (10.16 cm i.d.). Dark chambers were used to prevent heating effects from solar radiation. Because photosynthesis was also prevented by these dark chambers, reported CO_2 fluxes represent soil respiration rather than net ecosystem production. The chamber bottom consisted of a PVC pipe (5 cm long, 10.16 cm diameter) with a beveled edge to minimize soil disturbance; in several cases, a 7.62-cm-long base was used where there was standing water. Bases were installed approximately 30 min before sampling. Although this may have caused a small amount of disturbance to soil and root activity, bases were unable to be permanently installed due to homeowner concerns and periodic storm flows in ditches. Before collecting gas samples, a standard rubber band was placed around the PVC base. The chamber top was a PVC cap (10.16 cm i.d.). Two rubber septa were installed equidistant from the center of the lid; one was used for sampling, and a 21-gauge syringe needle was inserted through the other to serve as a vent (Hutchinson and Mosier, 1981).

Fig. 2. Photograph of example study ditch and adjacent road and lawn.

For a single gas flux measurement, the chamber top was mounted, and a syringe was inserted into the main septum to take an initial gas sample. Samples were injected into pre-evacuated 10-mL glass vials with butyl rubber septa. Vials were over-pressurized with injection of 15 mL gas to maintain the integrity of samples until analysis. Additional gas samples were taken from the chamber at 10, 20, and 30 min.

Samples were analyzed for N_2O and CH_4 via gas chromatograph. Methane separation was performed using a Carboxen 1006 PLOT capillary column (Supelco, Inc.) and a flame ionization detector. Carbon dioxide concentrations were analyzed on a LI-6200 Portable Photosynthesis System attached to an LI-6250 CO_2 Analyzer (LI-COR Inc.) during 2014. For 2015 samples, CO_2 was analyzed with the gas chromatograph flame ionization detector after installation of a methanizer, but calibration standards were from the same source as in 2014. Calibration and flux determination was as described for potential denitrification. Fluxes were converted from volumetric to mass-based units (μg gas m^{-2} h^{-1}) using the ideal gas law.

At the time of gas sampling, field measurements were also made to characterize soil conditions at each of the ditch and lawn sampling locations. A digital thermometer was used to record soil temperature. Soil moisture was assessed as volumetric water content (VWC) using a Hydrosense II (Campbell Scientific Inc.) probe.

Soil organic matter (OM) was assessed for dried, ground, and sieved (<2 mm) soil samples from all sites. Percent OM was quantified via loss-on-ignition at 500°C for 2 h (Ferguson and Swenson, nd). Percent C and N were quantified using a Carlo Erba NC2500 Elemental Analyzer (CE Elantech Inc.). Extractable soil NO_3^- was assessed using a deionized water extraction on fresh soil samples with gravel removed (Barrett et al., 2009), where NO_3^- was analyzed on a Dionex ICS-2000 Ion Chromatograph (Thermo Fisher Scientific Inc.). Soil pH was measured on a 1:2 soil and water slurry (Robertson et al., 1999).

Ditch water samples were taken during or immediately after six storm events between July 2014 and June 2015, including several storm events that directly preceded GHG sampling (Supplemental Table S3). Water samples were obtained in all ditches except one that never had observed flow. Additionally, stream water samples were obtained at least monthly from upstream (above the majority of ditches in the watershed) and watershed outlet locations during both storm and baseflow conditions (Fig. 1). Water samples were filtered to 0.45 μm and analyzed within 2 d for NO_3^- using a Dionex ICS-2000 ion chromatograph. Samples were acidified to pH 2 and analyzed for dissolved organic carbon (DOC) using an OI Analytical Total Carbon Analyzer Model 1010. Nitrate loads were also estimated in ditches to compare with extrapolated estimates of denitrification N removal capacity in ditches. Annual ditch NO_3^- load was calculated using the average NO_3^- concentration across all ditch sites and sample events, which was multiplied by annual ditch discharge (July 2014–June 2015). Ditch discharge was determined from total watershed stormflow because field observations indicated that the ditches in the study watershed only flowed during and directly after storm events. Total watershed stormflow was calculated using hydrograph separation as performed by the Web-based Hydrograph Analysis Tool (Engel, 2004). The stream discharge analyzed with the tool was obtained using a HOBO (Onset Computer Corp.) water level sensor at the watershed outlet and a stage–discharge rating curve developed during the study period using a Marsh-McBirney FloMate electromagnetic velocity sensor. To calculate the discharge contributed directly from the ditches, total watershed stormflow was scaled by the percentage of the watershed area that is drained by ditches.

To extrapolate gas flux measurements to the entire watershed, a detailed land use map (Fig. 1) was created in ArcGIS 10.3 using orthoimagery and ground surveying of ditches, which included measurement of width. Total watershed area for each land use was calculated. For the two land uses investigated in this study (ditches and lawns), areas were multiplied by average gas fluxes to generate area-weighted emissions. Emissions were expressed as CO_2 equivalents (CO_2eq) by multiplying by the 100 yr GWP of each gas. Potential denitrification was extrapolated in ditches by converting the rate to an areal basis using soil bulk density and multiplying the rate by total ditch area and a depth of 5 cm, the depth at which the denitrification assay was performed.

Statistical analyses were conducted in R v. 3.1.3 (The R Project for Statistical Computing, 2015). The two replicate sampling locations within each ditch or lawn site were averaged for all analyses. Given the skewed distribution of the gas fluxes, a paired Wilcoxon rank sum nonparametric test was used to assess differences between gas fluxes and potential denitrification at the ditch and lawn sites, where ditch and lawn measurements at a given location were paired. To assess any correlations between measured variables (including annual averages for gas fluxes, VWC, and ditch stormwater NO_3^- and DOC and one-time measurements for all other variables), Pearson product-moment correlation analysis was used. Additionally, temperature and moisture effects on GHG fluxes were assessed using linear mixed effects models (R packages lme4 and lmerTest) that were applied to transformed gas fluxes. For the models, lognormal or Box-Cox transformations were applied to fluxes with a constant added such that all fluxes for that gas were greater than zero, and the transformation was determined according to which improved normality the most. For temperature models, sites were treated as a random effect to account for having multiple time points from each site. For moisture models, sites and sample dates were treated as a random effect because moisture varied widely and because there was no consistent temporal trend for the sample dates. Normal distribution of residuals was graphically verified for all models. Significance for all tests was determined at $p < 0.05$.

Results and Discussion

Potential Denitrification

Potential denitrification was significantly higher in the ditches compared with the adjacent lawn sites (Fig. 3). Potential denitrification in the ditches was 3.45 ± 0.79 mg N kg^{-1} h^{-1} (mean ± SE). This is also greater than mean potential denitrification rates measured in various types of stormwater basins (0.2–2.5 mg N kg^{-1} h^{-1}) (Bettez and Groffman, 2012; McPhillips and Walter, 2015) and near the highest mean rates observed in stream riparian areas (0.4–3.6 mg N kg^{-1} h^{-1}) (Bettez and Groffman, 2012; Groffman and Crawford, 2003), which are other landscape features considered to be "biogeochemical hotspots." Thus, these ditches are potentially able to provide the valuable service of removing some excess N from stormwater before it pollutes downstream water bodies. Despite findings in other studies that denitrification is positively correlated to soil OM and moisture (Bettez and Groffman, 2012; Groffman and Crawford, 2003), in this investigation denitrification was not significantly correlated to either of these variables. The lack of correlation with OM in this study may be due to a narrower range of OM content (4–10%) than that observed in these other studies (~5–20%) (Bettez and Groffman, 2012; Groffman and Crawford, 2003). The only significant relationship was between denitrification rates in ditch sites and average DOC concentration in ditch stormwater (Table 1; Supplemental Table S4). This implies an influence of inflowing stormwater composition—particularly C availability—on nutrient cycling processes occurring in ditch soils or could present some reactive component of soil OM that is influencing nutrient cycling processes.

Greenhouse Gas Emissions

Methane emissions were significantly higher in ditches (624.8 ± 330.7 μg CH_4–C m^{-2} h^{-1}) compared with adjacent lawn sites (30.9 ± 17.7 μg CH_4–C m^{-2} h^{-1}) (Fig. 4a; Supplemental Table

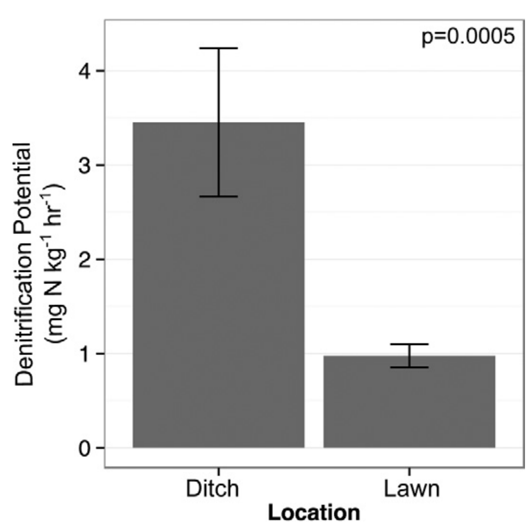

Fig. 3. Denitrification potential (mg N kg soil^{-1} h^{-1}) for soils within all ditches (*n* = 12) and adjacent lawns (*n* = 12) sampled in October 2014. Error bars denote SE. Reported *p* value represents result of a paired Wilcoxon rank sum test.

Table 1. Pearson correlation coefficients for measured variables. For measurements made more than once throughout the year, average values were used in the analysis.

	N_2O	CH_4	CO_2	Denitrification	Soil properties†						Inflow‡
					VWC	OM	N	C	pH	NO_3	NO_3
N_2O											
CH_4	0.20										
CO_2	0.47*	−0.34									
Denitrification	0.06	0.13	−0.12								
VWC	−0.21	0.61***	−0.49*	0.33							
OM	−0.30	−0.06	−0.12	0.13	0.24						
N	0.26	0.13	0.46*	0.27	0.16	0.33					
C	0.25	0.08	0.50**	0.16	0.08	0.28	0.91***				
pH	0.07	0.43*	0.00	−0.04	0.22	0.13	0.16	0.22			
soil NO_3	0.42*	-0.11	0.49*	−0.12	−0.30	−0.41*	0.04	0.18	0.17		
Inflow NO_3	0.53	0.13	0.01	0.33	−0.28	−0.22	0.35	0.14	−0.05	0.01	
Inflow DOC§	0.46	-0.01	0.55	0.65*	−0.12	0.20	0.35	0.27	0.10	0.45	0.25

* Significant at the 0.05 probability level.

** Significant at the 0.01 probability level.

*** Significant at the 0.001 probability level.

† OM, organic matter; VWC, volumetric water content.

‡ Inflow refers to inflowing ditch stormwater.

§ DOC, dissolved organic carbon.

S5). This trend was driven by higher average soil moisture in the ditches compared with the lawn sites (61 vs. 41% VWC). Across all sites, average CH_4 fluxes were significantly positively correlated with average soil VWC and pH (Table 1). Across all time points, the mixed effects model identified a positive correlation between CH_4 and VWC (Supplemental Table S6). The model also indicated that more variance was explained by site-to-site wetness differences (variance, 0.0008) than by temporal changes in moisture (variance, 0.0002). Methane emissions were particularly high (>500 μg C m^{-2} h^{-1}) in ditches where standing water was often observed and reached a maximum of 23,070 μg CH_4–C m^{-2} h^{-1}.

Methane emissions in ditch sites, particularly those with the highest soil VWC, were comparable to those measured in natural wetlands as well as wet stormwater detention basins. In natural fens in central New York and southeastern New Hampshire, mean fluxes of 734 to 5933 μg CH_4–C m^{-2} h^{-1} have been observed (Frolking and Crill, 1994; Smemo and Yavitt, 2006). In wet stormwater detention basins in central New York, mean fluxes of 2756 μg CH_4–C m^{-2} h^{-1} were observed (McPhillips and Walter, 2015). Methane emissions at the lawn sites were similar, but somewhat higher, than those observed in other lawn studies. Although some lawn studies have recorded slightly negative mean fluxes, reflecting net uptake or oxidation of CH_4 (Groffman and Pouyat, 2009; Kaye et al., 2004; Livesley et al., 2010), lawn sites in this study averaged 30.9 μg CH_4–C m^{-2} h^{-1}. This may be due to elevated average soil moisture in lawns in this study (average, 41% VWC) given that the sampling design targeted several dates directly after storm events. Moisture levels observed in other lawn studies range from around 10 to 40% VWC (Groffman and Pouyat, 2009; Livesley et al., 2010). Additionally, the relatively high CH_4 fluxes at these lawn sites may be due to reduced CH_4 uptake, which has previously been documented in urban and N-fertilized soils (Costa and Groffman, 2013; Groffman and Pouyat, 2009; Steudler et al., 1989).

For the other two GHGs of interest, the patterns were opposite to those of CH_4. Nitrous oxide emissions were significantly higher at the lawn sites (mean, 6.4 μg N m^{-2} h^{-1}) compared with the ditches (mean, 3.0 μg N m^{-2} h^{-1}) (Fig. 4b). Overall, N_2O fluxes were low except for a few "hot moments" of emissions at both the ditch and reference sites where emissions exceeded 50 μg N m^{-2} h^{-1}. These rates are similar to measurements made in lawns in Baltimore, MD, which ranged from −0.07 to 63 μg N m^{-2} h^{-1} (Groffman et al., 2009b; Raciti et al., 2011a), but are lower than fluxes measured in stormwater biofiltration basins in Australia (mean fluxes, 13.7 and 65.6 μg N m^{-2} h^{-1} for two basins) (Grover et al., 2013). Fluxes measured in this ditch and lawn study are also similar to those from lawns in California (median, 7.2 μg N m^{-2} h^{-1}), although much higher fluxes were observed from the California lawns after fertilization events (maximum, 720 μg N m^{-2} h^{-1}) (Townsend-Small and Czimczik, 2010). Higher fluxes may have occurred at the ditch and lawn sites, which were not captured given the limited number of chamber sampling events and episodic nature of N_2O emissions (Molodovskaya et al., 2012).

Across all sites, average N_2O fluxes were significantly positively correlated with soil NO_3^- and negatively correlated with soil VWC (Table 1). This is in line with previous findings on N_2O controls, which note that availability of NO_3^- and oxygen promotes production of N_2O during denitrification (Firestone et al., 1980). Nitrous oxide may also be produced by oxidation of NH_4^+ to NO_3^- in nitrification, which can occur at higher rates in lawns due to NH_4^+ availability and oxic soil conditions (Raciti et al., 2008, 2011b). Nitrous oxide fluxes were significantly higher in fertilized lawns compared with unfertilized lawns (Fig. 5a). There were no differences in N_2O fluxes from ditches grouped by fertilizer application (Fig. 5c). The combination of this lack of difference with the high denitrification potential and VWC in ditch soils indicates the possibility that denitrification is proceeding to N_2 in the wetter ditch sites. Additionally, ditches are experiencing influence from both the fertilization of the adjacent

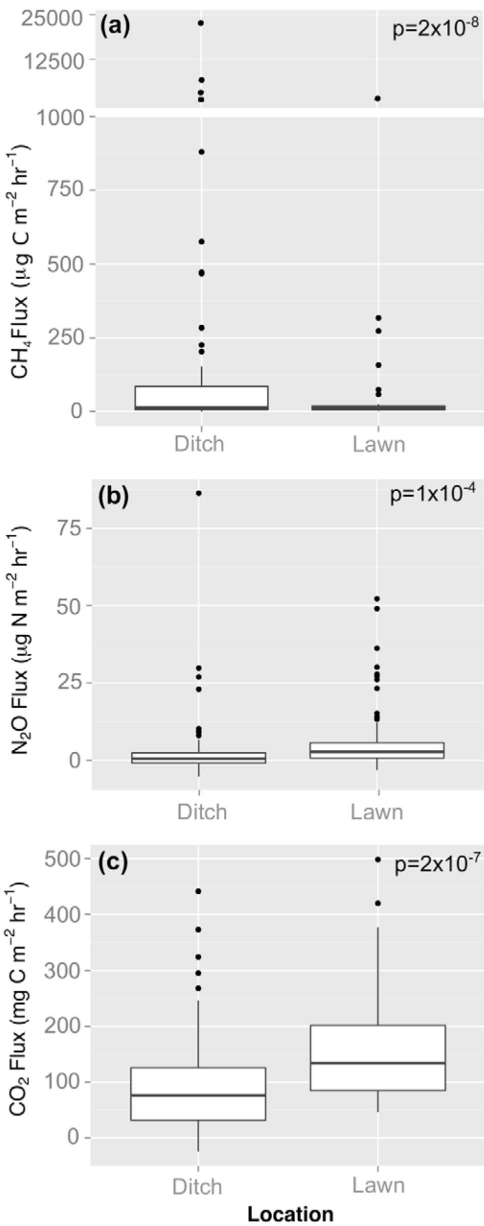

Fig. 4. Boxplots of (a) CH_4 fluxes, (b) N_2O fluxes, and (c) CO_2 respiratory fluxes for all sites. Data include all 12 ditch or lawn locations for all six time points. Horizontal black lines show the data median, the box boundaries mark the first and third quartiles, the whiskers indicate 1.5 times the interquartile range, and the points represent extreme values. Reported *p* values represent results of paired Wilcoxon rank sum tests.

lawns as well as the runoff from the rest of the ditch's watershed, confounding the influence of these factors.

Carbon dioxide respiratory emissions were statistically significantly higher from lawn sites compared with ditch sites (Fig. 4c). Average fluxes at all sites (lawn, 151 mg C m^{-2} h^{-1}; ditches, 98 mg C m^{-2} h^{-1}) were comparable to those measured from lawns in the similar climatic conditions of the Boston, MA, region (mean, 194 mg C m^{-2} h^{-1}) (Decina et al., 2016) and were lower than those reported for lawn sites under a range of management practices in the temperate coastal climate of Melbourne, Australia (>300 mg C m^{-2} h^{-1}) (Livesley et al., 2010). Across all sites, average CO_2 fluxes were significantly positively correlated with soil NO_3^- and % soil C and N. Carbon dioxide was also significantly negatively correlated with soil VWC (Table 1); this relationship is likely driving the lower respiratory emissions measured here compared with similar studies, given that this investigation targeted several dates after storm events and thus conditions were wetter. Across all time points, the mixed effects model identified a significant positive correlation between CO_2 fluxes and soil temperature, which concurs with broadly observed relationships between temperature and soil respiration (Kirschbaum, 1995). As with N_2O, CO_2 fluxes were significantly higher in fertilized than in unfertilized lawns (Fig. 5b), suggesting increased ecosystem respiration with increased N availability. Carbon dioxide fluxes were also significantly higher in ditches located adjacent to fertilized lawns compared with unfertilized lawns (Fig. 5d). Increased respiratory CO_2 production with fertilizer addition has been observed in other urban lawn and garden systems (Livesley et al., 2010) and may be driven by increased primary production from nutrient inputs (Kaye et al., 2005). Thus, despite the increase in CO_2 emissions from ecosystem respiration, there also may be a corresponding increase in primary production leading to net C sequestration in these fertilized lawns. Net primary production quantified for turfgrass in Maryland was not found to differ with fertilization (Lilly et al., 2015). Although respiration was not quantified in that investigation, the findings imply that any changes in respiration due to fertilization were countered by changes in gross primary production. Primary production was not quantified in this study, although C content of fertilized lawn and ditch soils was greater than their nonfertilized counterparts, implying that fertilized sites may be storing C. In contrast, OM content did not differ with fertilization (Supplemental Table S1).

Examining the total GHG fluxes from ditches and lawns in this study, emission rates from lawns are greater (Fig. 6a). Despite higher rates of CH_4 emissions in ditches, the higher rate of N_2O and CO_2 emissions from lawns combined with the greater GWP of N_2O (298 times greater) relative to CH_4 (25 times greater) makes lawns the greater emitters on a mass flux per area basis. If only N_2O and CH_4 are considered, ditches do have a greater rate of GHG emissions per area (Fig. 6a, zoomed-out panel).

Extrapolation up to the Watershed

Within this study watershed, forest was the dominant land type (50.36% of total watershed area); lawns occupied 29.91% of the total watershed area, and ditches occupied 0.74%. Combining the total area with the averaged GHG fluxes, lawns in the watershed emitted 1179 Mg CO_2eq yr^{-1}, compared with 22 Mg CO_2eq yr^{-1} from ditches (Fig. 6b). Thus, even though ditches had higher rates of CH_4 emissions, the greater total area occupied by lawns combined with their greater rates of N_2O and CO_2 emissions made them much greater overall emitters of GHGs. Some of these emissions are likely offset by C sequestration in lawn and ditch soils. Previous work in lawns has identified substantial C sequestration (Campbell et al., 2014; Townsend-Small and Czimczik, 2010), and a study of vegetated swales along highways found increasing soil C density with age (Bouchard et al., 2013).

Lawn fertilization strongly affected the overall GHG footprint of lawns in this investigation. The estimate of 1179 Mg CO_2eq yr^{-1} used the average flux from all 12 sites, which is reasonable given that 43% of homeowner surveys in the

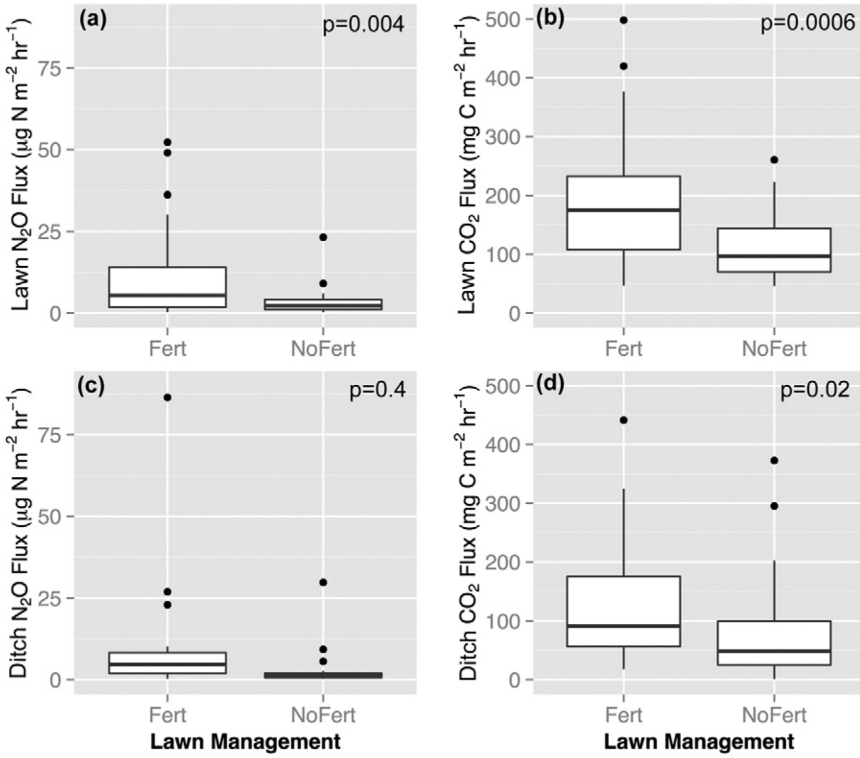

Fig. 5. Boxplots of (a) N_2O fluxes and (b) CO_2 respiratory fluxes from fertilized or unfertilized lawns and (c) N_2O fluxes and (d) CO_2 respiratory fluxes from the ditches adjacent to these lawns for all six time points. Horizontal black lines show the data median, the box boundaries mark the first and third quartiles, the whiskers indicate 1.5 times the interquartile range, and the points represent extreme values. Reported *p* values represent results of Wilcoxon rank sum tests.

watershed (n = 28) indicated that they applied lawn fertilizer. If emissions were upscaled using only average fluxes from the fertilized or nonfertilized sites, lawns emitted 889 Mg CO_2eq yr^{-1} if all were assumed to not be fertilized, whereas emissions increased 65% to 1470 Mg CO_2eq yr^{-1} if all lawns were assumed to be fertilized. It is uncertain how these GHG emissions under different management scenarios would be countered by primary production.

For denitrification, extrapolation of potential denitrification rates to the full watershed network of ditches yielded a maximum N removal potential of 42,805 kg N yr^{-1}. This can be compared with the ditch NO_3^- load, which was calculated as 118 kg N yr^{-1}. This is a coarse estimate of NO_3^- load based on a relatively small number of water samples and a simple model of ditch flow, which assumed that ditch contributions to stream stormflows scale with ditch area. Uncertainty in this relationship could come from factors such as differences in how ditch-drained and non–ditch drained portions of the watershed contribute to stream baseflow versus stormflow. Despite this uncertainty, the much greater magnitude of potential denitrification demonstrates that the ditches could provide more than enough removal of N loads in ditch stormwater.

There are several key assumptions in the calculations of denitrification N removal that are important to consider. This rate is representative of maximum denitrification under optimal conditions (e.g., anaerobic, ample N and C). The actual in situ rate likely varies throughout the year based on temperature, wetness, etc. (Groffman et al., 2009a). Additionally, N removal was calculated for a ditch soil depth of 5 cm. With the high velocity of water movement during storms in these ditches, much of the incoming N may not have an opportunity to interact with ditch soils beyond the sediment surface, although there may also be denitrification occurring in soil water or groundwater at depth in the lawns and ditches.

Fig. 6. Average greenhouse gas fluxes for ditches and lawns on (a) a per area basis and (b) scaled by watershed area. Fluxes are represented in CO_2 equivalents according to their global warming potential. Zoomed out panels show N_2O and CH_4 data only.

Despite the high ditch denitrification potential, increases in average NO_3^- from 0.27 to 0.38 mg N L^{-1} (Supplemental Table S4) between upstream and outlet stream sites indicate that the developed, ditch-drained portion of the watershed continued to contribute some N. One key limitation to reaching this potential N removal is residence time of N in ditches because observations during storms indicate that most stormwater is moving at high velocity and may not have an opportunity to infiltrate or interact sufficiently with soils and the associated microbial communities. Research in stream sediments has found that residence time of water and associated substrates is a key positive influence on denitrification in the subsurface hyporheic zone (Zarnetske et al., 2011). Future research should investigate ditch denitrification rates in situ and quantify ditch water residence time and exchange with ditch soils.

Management Implications

These findings have several implications for management of suburban landscapes. For lawns, it is clear that nutrient availability is a driver of GHG emissions, particularly N_2O, and this nutrient availability is linked to fertilizer application. Carbon dioxide respiratory emissions also increased at fertilized sites, although it is uncertain whether these sites would be net C sources or sinks because fertilization could also increase primary production. Working with homeowners to better optimize fertilizer application to lawns in terms of amount and timing could help reduce excess nutrients available for transformation into GHGs or runoff into ditches.

In managing ditches, goals should include optimizing water quality services like denitrification while reducing emissions of CH_4. Methane emissions were highest from the wettest ditches, particularly those with standing water. From observations, standing water occurred when metal culverts under driveways were blocked by debris or were not flush with the ditch. Thus, maintenance practices, such as using PVC culverts in place of metal, could prevent warping that might back up water, and grates or screens placed over culvert ends could prevent blockages caused by accumulation of debris.

We found that grassed roadside ditches were hotspots for potential denitrification that can prevent excess NO_3^- from reaching downstream water bodies. Although it was not clear which environmental factors most strongly promoted potential denitrification at these sites, it is clear that these ditches overall are hotspots for this process relative to lawns, which dominate suburban landscapes. Additionally, extrapolated estimates of denitrification indicated that there is potential for these ditches to remove all incoming NO_3^- in stormwater. However, factors, such as the limited residence time of stormwater, are preventing ditches from reaching this potential.

Managers of suburban watersheds should consider how to modify ditch design to increase interaction of stormwater with ditch soils to promote biogeochemical processing in ways that would not simply back up water for extended periods and promote CH_4 emissions. One existing management practice that inherently reduces interactions with soils is the practice of scraping ditches every few years to reduce the risk of ditches overflowing into the road during large storm events (Johnson, 2015). Although this practice did not occur at our study sites and thus was not directly evaluated, we predict that the removal of soil and vegetation would reduce the beneficial NO_3^- removal potential observed in our ditches, and therefore stopping scraping could improve water quality benefits. Previous work in grassed highway swales in Maryland found that adding check dams reduced NO_3^- loads in runoff, likely due to infiltration and denitrification (Stagge et al., 2012). In agricultural settings, modification of ditches to be "two-stage," which means adding a secondary floodplain to allow increased soil contact during high flows, enhanced overall N removal (Roley et al., 2012). For suburban ditches, this approach could be modified such that ditches could overflow laterally onto a portion of the adjacent lawn. Other management options to increase opportunities for water quality improvement include strategically placed stormwater bioretention basins (e.g., where there is available space to route ditch water into a basin for longer detention) or subsurface woodchip bioreactors in ditches. Bioreactors are units filled with a C source (e.g., woodchips) and have been used to promote NO_3^- removal via denitrification in agricultural tile drainage (Addy et al., 2016) but could be implemented to remediate nutrient-rich water in other settings as well.

Conclusions

This study provides the first known data on GHG emissions and potential denitrification in grassed roadside ditches in a suburban landscape. With their pulses of stormflow and associated solutes and particulates, ditches have the potential to be important biogeochemical hotspots in developed landscapes. Comparing these features with adjacent lawns in the study watershed, we found that ditches were hotspots for CH_4 emissions due to higher wetness but that lawns had greater rates of N_2O emissions and CO_2 respiratory emissions. Fertilization and nutrient availability influenced N_2O and CO_2 fluxes, driving greater emissions for both gases from fertilized versus unfertilized lawns and greater CO_2 respiratory emissions in ditches adjacent to fertilized lawns. Extrapolating GHG emissions up to the watershed, the much greater area of lawns relative to ditches made lawns the overall greater emitter of GHGs, although primary production (not measured here) does have the potential to counter some or all of these emissions. Potential denitrification rates were greater in ditches, demonstrating that ditches can provide important N removal services. Future research should focus on identifying design and management strategies to minimize CH_4 emissions in ditches while promoting beneficial processes like denitrification.

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