Road ditches in the suburban landscape: potential hotspots for nutrient transport and cycling

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**Key Points**

- Water quality and nutrient cycling processes were measured in grassed road ditches in a suburban watershed.
- Ditches were sources of nitrogen and phosphorus. Only increases in nitrate were observed in the receiving stream, though nitrate concentrations are still low compared to agricultural streams in the region.
- Despite being sources of nitrate, ditch soils had high denitrification rates, which indicate potential for soil microbes to remove excess nitrate.
- Wet ditches were sources of methane, but adjacent lawns were greater emitters of nitrous oxide and carbon dioxide.
- Better management of lawn fertilizer application and minimizing standing water in ditches could reduce detrimental environmental impacts in suburban landscapes.
Introduction

Urban and suburban areas are a growing fraction of the landscape in the United States, increasing in area by 400% in the past 50 years (Brown et al., 2005). With the infrastructure and development that comes with this land use change, there is also significant manipulation of hydrology (Davis and McCuen, 2005). Construction of buildings and parking lots necessitates stormwater retention basins or other management practices and, with roads, comes the building of ditches to route water quickly away (Buchanan et al., 2013).

In design of road ditches, there is often no consideration given to how this routing of runoff might impact downstream water quality. Recent research in an agricultural watershed in central New York found that road ditches served as an ‘efficient conduit’ of sediment, phosphorus, and E. coli to receiving streams (Buchanan et al., 2013; Falbo et al., 2013). While there is a substantial body of research on contaminant loads in road runoff (e.g. Miguntanna et al., 2013; Zhang et al., 2013), there has been minimal research on the role of road ditches in water quality, particularly with regards to nitrogen (N). In suburban watersheds, which often have a high density of roads and corresponding ditches, potential N sources to ditches include fertilizer applied to residential and commercial lawns (Raciti et al., 2011b) as well as automobile exhaust and atmospheric deposition (Bettez and Groffman, 2013). With excess loads of N causing water quality impairment of estuaries (Seitzinger and Sanders, 1997), it is essential that we work to minimize its export from streams and rivers where possible.

Road ditches are constructed of a variety of materials, ranging from concrete to more natural vegetated soil ditches (US EPA, 2000). With the nature of concrete as a relatively impermeable and unreactive substance, once nutrients or other pollutants enter a concrete ditch, they are likely to be transported to whatever water body receives the ditch outflow. However, vegetated soil ditches may provide some opportunity for nutrients that enter the ditch to be transformed and/or removed due to biological or chemical processes before reaching the end of the ditch. In particular, denitrification, a microbiologically-mediated process that transforms nitrate (NO$_3^-$) into nitrous oxide (N$_2$O) or dinitrogen (N$_2$) gas (Tiedje, 1994), may be occurring in vegetated ditches and acting as an important nitrate removal pathway. Previous research has documented the occurrence of denitrification hotspots in the environment where there is the convergence of nutrient inputs and varying moisture and oxygen conditions (Groffman et al., 2009a; McClain et al., 2003)- conditions which are likely satisfied in road ditches.

Some of these conditions which promote denitrification (e.g. high soil moisture and soil carbon) also influence production of certain greenhouse gases. Nitrous oxide (N$_2$O), a greenhouse gas with 298 times the warming potential of carbon dioxide (CO$_2$), can be a by-product of denitrification. Additionally, methane (CH$_4$) is a greenhouse gas with 25 times the warming potential of CO$_2$ (Solomon et al., 2007), which is produced by soil microbes when there is high soil moisture and ample soil carbon.

Despite the fact that ditches parallel the majority of roads in New York State (Buchanan et al., 2013), we have limited understanding of their environmental impacts. Particularly in suburban areas of the state, a land use type experiencing substantial growth (Roberts, 2006), road ditches could be an important contributor of pollutants and excess nutrients to streams and rivers as well as a source of greenhouse gases. Thus, it is necessary to assess whether ditches are indeed having any of these negative
impacts and identify whether there are certain ditch designs or landscape management practices that could reduce these impacts.

**Objectives**

The objectives of this study were to (1) quantify denitrification and greenhouse gas emissions in twelve grassed ditches, (2) link any observed patterns in biogeochemical processes to ditch design or landscape characteristics, and (3) survey nutrient concentrations in ditch stormwater and streamwater to determine whether ditches are contributing to downstream impairment.

**Methods**

**Study Site**

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) that empties into Cayuga Lake. The 332 ha mixed use watershed is representative of many suburban watersheds across New York State, with 40% urban (including residential, school, and commercial land), 44% forest, 8% open water or wetland and 8% pasture. Existing stormwater infrastructure in the watershed includes detention ponds, road ditches (almost all are vegetated), and minimal storm sewers, all of which discharge within the watershed (Easton and Petrovic, 2008).

Potential experimental sites were identified through use of a homeowner survey. Information was mailed to homes which had ditches that drained within the study watershed. Homeowners who were interested in allowing their ditch to be a research site were directed to an online survey form. The survey was granted an exemption from the Cornell Institutional Review Board as it did not solicit personal information or pose a risk to participants. The survey solicited information on lawn maintenance (including mulching and fertilizer application) as well as road ditch conditions (such as how often it had water flowing). Using this information, twelve grassed road ditches were selected as experimental sites, such that they represented a range of lawn care practices and ditch moisture regimes.

![Figure 1. Map of stream and ditch sampling sites used for water quality assessment](image-url)
Water Quality Data Collection

Samples for water quality analysis were taken from the ditch research sites as well as three sites along the main stream (Figure 1). Ditch water samples were only taken during or immediately following storm events, since this was the only time that most of the ditches had standing or running water. Samples were taken during four storm events in summer and fall 2014. At each ditch which had water, a 125 mL HDPE bottle was filled. We were unable to obtain water samples at one ditch due to lack of water during all storm events. Stream water samples were taken during the summer, fall, and winter in both baseflow and stormflow conditions, to capture the full range of nutrient concentrations. Upon return to the lab, all water samples were filtered and stored at 4°C until water quality analysis. Each sample was analyzed for NO\textsubscript{x} (NO\textsubscript{3}\textsuperscript{-} + NO\textsubscript{2}\textsuperscript{-}) and Cl using a Dionex ICS-2000 Ion Chromatograph. Dissolved organic carbon (DOC) was analyzed using an OI Analytical Total Carbon Analyzer Model 1010 and soluble reactive phosphorus (SRP) was assessed with an OI Analytical Phosphorus Analyze Model FS3000.

Soil Greenhouse Gas Emissions

Emissions of CO\textsubscript{2}, CH\textsubscript{4} and N\textsubscript{2}O were measured in the ditches on four dates between July and October 2013 that spanned a range of temperature and moisture conditions. At each ditch, flux measurements were made at two locations within the ditch and two reference locations adjacent to the ditch (Figure 2). Fluxes were measured using in situ static PVC chambers (4 in. i.d.). The chamber bottom consisted of a 2 in. long, 4 in. diameter PVC pipe with a beveled edge; in several cases, a 3 in. long base was used where there was standing water. Bases were installed immediately prior to sampling but the beveled edge minimized soil disturbance. Prior to measurement, a standard rubber band was placed around the PVC base. The chamber top was a PVC cap (4 in. 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).

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 overpressurized with injection of 15 mL gas in order to maintain the integrity of samples until analysis. Additional gas samples were taken from the chamber at 10, 20, and 30 minutes.

Samples were analyzed for N\textsubscript{2}O and CH\textsubscript{4} using an Agilent 6890N gas chromatograph equipped with a HP 7694 Headspace Autosampler (Hewlett-Packard Co.). N\textsubscript{2}O separation was performed using a Supel-Q™ PLOT capillary column (30m x
0.32mm; Supelco Inc.) with ultra-pure helium carrier gas (2.6 mL min\(^{-1}\)) and 95:5 Ar:CH\(_4\) make-up gas (8.2 mL min\(^{-1}\)) and a µECD (electron capture detector) set to 250°C. CH\(_4\) separation was performed using a Carboxen 1006 PLOT capillary column (30 m x 0.32mm; Supelco, Inc.) and an FID (flame ionization detector) set to 200°C with H\(_2\) gas (30 mL min\(^{-1}\)), air (400 mL min\(^{-1}\)), and N\(_2\) makeup gas (25 mL min\(^{-1}\)). Oven temperature was initially set to -22°C for 4.7 min, then increased to 30°C for 2.3 min to allow for elution of both gases of interest. CO\(_2\) concentrations were analyzed on a LI-6200 Portable Photosynthesis System attached to an LI-6250 CO\(_2\) Analyzer (LI-COR, Lincoln, NE). Calibration curves were made using serial dilutions of 1 ppm N\(_2\)O, 20 ppm CH\(_4\), and 1000 ppm CO\(_2\) (Airgas Inc.) Gas fluxes were calculated by determining the linear slope of the concentrations of the four time-points (Rochette and Bertrand, 2008). Fluxes were converted from volumetric to mass-based units (ug gas m\(^{-2}\) hr\(^{-1}\)) 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 reference sampling locations. A thermometer was used to record soil temperature. Soil moisture was assessed as volumetric water content (VWC) using a Campbell Scientific Hydrosense II probe. Electrical conductivity (EC) was measured using a Spectrum Field Scout Soil EC probe.

**Soil Characterization and Denitrification Potential**

During summer 2014, soil cores were extracted from each ditch and reference sampling site for measurement of soil properties; at each site three 5 cm cores were removed and homogenized. Soil pH was measured using a 1:2 soil: water solution (Robertson et al. 1999). Soil organic matter (OM) was measured via loss-on-ignition (Ferguson and Swenson, n.d.). A second set of cores was taken for assessing potential denitrification, which is the maximum capacity of a soil microbial community to denitrify. The cores were processed according to the denitrification enzyme assay protocol (Groffman et al., 1999). Approximately 5 g soil subsamples were incubated in 125 mL bottles under anaerobic conditions with ample NO\(_3^-\) and carbon (C) substrate. Chloramphenicol was not used due to its adverse effects on existing denitrification enzymes (Pell et al., 1996). Acetylene gas was added to prevent conversion of N\(_2\)O to N\(_2\) gas; thus denitrification rate could be measured from production of N\(_2\)O between multiple time-points (0, 20, 40, and 60 minutes). Samples were stored and analyzed as previously noted. N\(_2\)O fluxes were determined using the
linear slope of the concentrations of the four time-points and converted to denitrification rates in mg N kg\(^{-1}\) hr\(^{-1}\) using the dry mass of the soil subsamples.

**Statistical Analysis**

Statistical analyses were conducted in R v. 3.1.2 (The R Project for Statistical Computing). A Wilcoxon rank sum non-parametric test was used to assess differences between gas fluxes and potential denitrification at the basin and reference sites. Analysis is ongoing and future assessment will focus on assessing differences in these biogeochemical processes based on landscape characteristics. Additionally, Pearson correlation analysis will be used on the entire dataset to discern what environmental factors (such as ditch water nutrient concentrations) may be influencing observed patterns.

**Results & Discussion**

Average NO\(_x\) in ditch stormwater ranged from 0.21 to 1.34 mg N L\(^{-1}\). The highest NO\(_x\) concentrations were observed in ditches where adjacent lawns were fertilized; however, additional landscape factors remain to be analyzed which may also play a role in these patterns. Average SRP in ditch stormwater ranged from 0.05 to 0.30 mg P L\(^{-1}\). Comparison between the three stream sampling sites indicated minimal change in SRP and overall low SRP concentrations (~0.03 mg P L\(^{-1}\)). However, NO\(_x\) exhibited an increase in average concentration from upstream (0.27 mg N L\(^{-1}\)) to midstream (0.50 mg N L\(^{-1}\)) followed by a slight decrease from midstream to downstream (0.38 mg N L\(^{-1}\)). The initial increase is likely due to NO\(_x\) inputs from the ditches. Whereas the upstream site has undeveloped headwaters, the watershed of the midstream site is characterized by substantial residential development and accompanying road ditches (Figure 1). Given that the watershed does not have septic systems or a wastewater treatment plant, ditches are a likely source of nutrients. Though more ditches continue to discharge into the stream between the midstream and downstream sites, the slight decrease in NO\(_x\) indicates that another process is countering contributions from the ditches. It is possible that groundwater with low NO\(_x\) concentrations is discharging into the stream bed in wetlands near the downstream site, or denitrification may be occurring in stream sediments. Slight decreases in Cl (a compound that is not biologically active) between midstream and downstream imply that some dilution of stream solutes is occurring due to groundwater inputs.

A U.S. Geological Survey nationwide study of nutrients in streams found that mean concentrations of SRP were 0.01 mg P L\(^{-1}\) in undeveloped streams and 0.03 mg P L\(^{-1}\) in urban streams, though some urban streams reached ~0.7 mg P L\(^{-1}\) (Dubrovsky et al., 2010). Thus, even though some of the ditches discharged water with higher P concentrations, our study stream was only slightly elevated above background levels. Mean concentrations of NO\(_3^-\) in undeveloped streams are 0.24 mg N L\(^{-1}\) and ~0.7 mg N L\(^{-1}\) in

*Figure 4. Potential denitrification summarized for all ditch and reference sites*
urban streams. Thus, downstream portions of our study stream do have elevated NOx compared to undeveloped watersheds, likely due in part to ditch contributions.

Potential denitrification was significantly higher in the ditches compared to the adjacent reference sites (p<0.05; Figure 4). Average potential denitrification in the ditches was 3.45 mg N kg\(^{-1}\) hr\(^{-1}\), which is comparable to values measured in stormwater detention basins and stream riparian areas- landscape features considered to be ‘biogeochemical hotspots’ (Bettez and Groffman, 2012; McPhillips and Walter, In prep). Thus, these ditches may be able to provide some removal of excess nitrogen in stormwater before it reaches nearby streams.

Greenhouse gas emissions were quite variable over the sampling dates, based on changes in temperature and moisture. However, some patterns stood out amongst the variation. CH\(_4\) emissions were significantly higher in ditches compared to adjacent reference sites (p<0.05; Figure 5b). This trend was likely driven by higher average soil moisture in the ditches compared to the reference sites (60% vs 43% VWC). CH\(_4\) emissions were particularly high (>0.5 mg C m\(^{-2}\) hr\(^{-1}\)) in sites where standing water was often observed; rates at these sites were comparable to those measured in natural wetlands (Frolking and Crill, 1994; Smemo and Yavitt, 2006).

For the other two gases of interest, the patterns were opposite. Given that these gases are produced by microbes in metabolic processes that require some presence of oxygen, it is not surprising that rates were higher at the reference sites which had less soil pore space filled with water. Average N\(_2\)O emissions were slightly higher at the reference sites (6.9 µg N m\(^{-2}\) hr\(^{-1}\)) compared to the ditch (3.4 µg N m\(^{-2}\) hr\(^{-1}\); Figure 5a), though the difference was not statistically significant. Overall, N\(_2\)O fluxes were low except for a few ‘hot moments’ of emissions at both the ditch and reference sites where emissions reached 80 µg N m\(^{-2}\) hr\(^{-1}\). These rates are comparable to measurements made in lawns in Baltimore, Maryland which ranged from -0.07 to 63 µg N m\(^{-2}\) hr\(^{-1}\) (Groffman et al., 2009b; Raciti et al., 2011a). Higher emissions have been observed in fertilized lawns, up to 720 µg N m\(^{-2}\) hr\(^{-1}\) (Townsend-Small and Czimczik, 2010).

Figure 5. Greenhouse gas emissions in all ditch and adjacent reference sites
CO₂ emissions were statistically significantly higher from reference sites compared to ditch sites (p<0.05; Figure 5c). Average fluxes at all sites (reference= 143 mg C m⁻² hr⁻¹, ditches= 80 mg C m⁻² hr⁻¹) were similar to those measured in stormwater bioretention cells (Grover et al., 2013) and were lower than those reported for lawn sites under a range of management practices (>300 mg C m⁻² hr⁻¹; Livesley et al., 2010).

**Conclusions & Management Implications**

This study evaluated nutrient cycling in grassed road ditches in a suburban watershed. A water quality survey was conducted for twelve ditches and the stream to which they drained. While some ditches had higher SRP concentrations, the stream's levels remained very low. The stream did exhibit an increase in NOₓ concentration from its headwaters to downstream developed areas, which likely was due to contributions from ditches. Though the ditches do seem to be serving as a source of NOₓ, there is evidence of N removal occurring in ditch soils, as potential denitrification was very high in the ditches.

Since C and N availability and variable hydrologic conditions can create a unique nutrient cycling environment in ditches, data was also collected on emissions of soil greenhouse gases. Some wet ditches were a source of CH₄, though CO₂ and N₂O were both higher in adjacent lawns than in ditches.

Application of lawn fertilizer is likely influencing ditch nutrient concentrations as well as emission of some greenhouse gases, both in the ditch and adjacent lawn. Homeowners should minimize fertilizer application and take care to optimize any additions for plant requirements. Additionally, designing ditches such that they do not retain standing water would help reduce CH₄ emissions.

This research is ongoing and further data collection and analysis will help elucidate patterns and what landscape factors are driving them.

**Outreach & Presentations**

Presentations of this work will be given at the International Association of Great Lakes Research Annual Conference in Burlington, VT in May 2015 and at the Ecological Society of America Annual Conference in Baltimore, MD in August 2015. A manuscript is in development for publication in a peer-reviewed journal, and a video is in progress to help disseminate our findings to the greater community.

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Abbreviations

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<tr>
<th>Abbreviation</th>
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<tr>
<td>DOC</td>
<td>Dissolved organic carbon</td>
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<tr>
<td>N</td>
<td>Nitrogen</td>
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<tr>
<td>NOx</td>
<td>$\text{NO}_3^- + \text{NO}_2^-$</td>
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<td>P</td>
<td>Phosphorus</td>
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<td>SRP</td>
<td>Soluble reactive phosphorus</td>
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References


