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The impact of pollutants from a major northern highway on an adjacent hardwood forest☆

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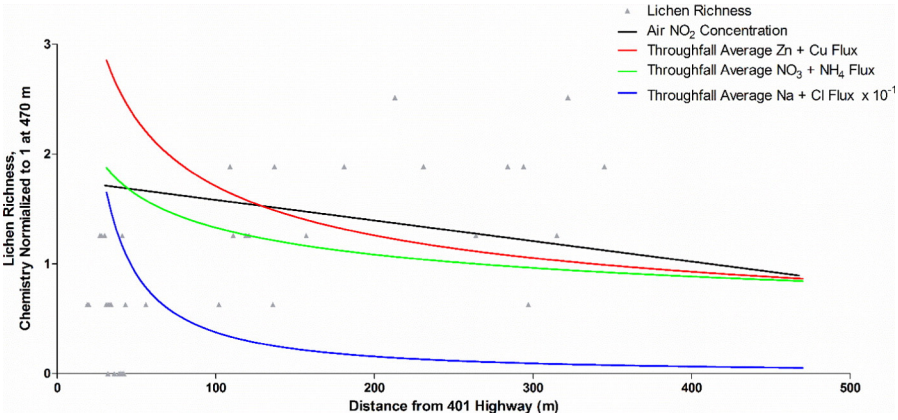
HIGHLIGHTS

- Inorganic N, base cations and metal deposition are elevated close to a busy highway.
- Pollution gradient is evident in epiphytic lichen chemistry.
- No impact of N or road salt on soil chemistry.
- Altered lichen and plant communities are evident up to 80 m from the highway.

GRAPHICAL ABSTRACT



Epiphytic lichen richness and normalized annual pollution deposition with distance from a major northern highway.

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ABSTRACT

Emissions of pollutants from highways can exert multiple stresses on adjacent ecosystems. In this study air concentrations of NO_2 and throughfall deposition of inorganic N (NO_3^- and NH_4^+), SO_4^{2-}, Cl^-, base cations and several metals were measured in all four seasons along a 1.5 km hardwood forest gradient extending away from a major highway (Highway 401) in southern Ontario, Canada. Soil and lichen chemistry and herbaceous plant and epiphytic lichen species composition were measured within the hardwood forest to evaluate impacts of these pollutants. Air concentrations of NO_2 and deposition of inorganic N, Cl^-, base cations and Cu and Zn in throughfall were significantly elevated within 100 m of the road compared with the more distant sites. Concentrations of several pollutants including N (and $\delta^{15}N$), Na^+, Al and Fe in epiphytic lichen tissue decreased with distance from the highway, and epiphytic lichen richness was lower at sites within 100 m of the road. Despite high throughfall inputs of >15 kg N ha^{-1} y^{-1} and 100 kg Na^+ ha^{-1} y^{-1} within 33 m of the highway, for example, there was no significant difference in soil chemistry amongst sites. Plant community composition at sites within 80 m of the highway differed from sites located further from the road, but it is unclear whether differences were due to highway emissions or were a result of natural forest edge effects.

☆ Capsule "major northern highways are a source of pollution that exert ecological impacts on adjacent ecosystems"

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1. Introduction

Studies have shown that exposure to pollutants emitted from highways can be extremely high close to busy roads, but that ecosystem exposure generally decreases rapidly with distance, with particulate pollutants such as metals and road salts decreasing more rapidly than gaseous pollutants such as nitrogen dioxide (NO_2) (Bell and Ashenden, 1997; Cape et al., 2004; Viskari et al., 1997). While ecological impacts close (<10 m) to highways can be quite severe, adverse impacts on vegetation communities or tree health have been observed at distances up to 200 m (depending on wind direction and traffic density) (Bernhardt-Römermann et al., 2006; Bignal et al., 2007). Nitrogen- and salt-tolerant species can out-compete other species (Truscott et al., 2005), and N-tolerant species abundance has been found to be correlated with the N deposition gradient (Lee and Power, 2013; Lee et al., 2012). Higher grass abundance has also been observed near roads as compared to farther away (Lee et al., 2012; Angold, 1997), which is correlated with N deposition gradients (Lee et al., 2012). A decrease in the abundance and health of lichens and bryophytes closer to roads has also been found (Angold, 1997) possibly due to exposure to high atmospheric NO_2 concentrations (Bignal et al., 2008).

Bignal et al. (2008) argued that further research needs to be conducted in different habitat types and climatic regions in association with different road types and pollution climates. There have been few detailed studies that have investigated the ecological impacts of motor vehicle emissions in North America; while several studies have documented increased N exposure close to roads these have rarely been linked to biotic impacts (Redling et al., 2013). Redling et al. (2013) demonstrated that NO_2 flux and $\delta^{15}N$–NO_2 values were significantly higher close to the road indicating a high proportion of automobile-sourced N is deposited near roadways. In addition, plant tissue $\delta^{15}N$ values were higher near the road, signifying the influence of automobile emissions on plant tissue composition. Watmough et al. (2014) demonstrated that springtime concentrations of NO_2 and NH_3 in southern Ontario forests could be predicted by road density and that epiphytic lichen richness was significantly negatively related to modelled NO_2 and ammonia (NH_3) exposure suggesting that roads may be having a widespread impact on forest ecosystems. Furthermore, the relative impact of N (NO_2 and NH_3) compared with other pollutants such as metals or road salt, which is increasingly used in northern regions is unclear. Bae et al. (2016) suggested that germination and growth of plants may be adversely impacted by high levels of heavy metals in roadside soils in Quebec. Similarly, road salt can impact plant species directly or it can impact plant communities indirectly through changes in soil chemistry or soil structure (Fan et al., 2014; Green et al., 2008; Heintzman et al., 2015).

In southern Ontario, the most populous Canadian province, total road length increased from 7133 km to 35,637 km in 60 years (Fenech et al., 2000). These numbers are expected to continue to grow in the future and contribute to air quality issues, although the impacts on adjacent ecosystems are uncertain. To date there have been no studies that have assessed the magnitude and potential ecological impact to forest ecosystems of multiple pollutants that may be emitted from northern highways in Canada. The objectives of this study therefore were to: 1) characterize differences in atmospheric NO_2 concentrations and pollutant loadings in throughfall with distance from a major highway in southern Ontario over a full year to capture seasonal differences, 2) assess the response of epiphytic lichen and ground flora species composition as well as soil and lichen chemistry along the same transect.

2. Methods

2.1. Study area

The study site is located south of Highway 401 just east of Coburg, Ontario and is approximately 3 km north of Lake Ontario (Fig. 1). The long-term (30-year) average temperature is 6.8 °C and the average annual precipitation is 986 mm. The forest is characteristic of the Great Lakes St Lawrence forest region with Orthic Melanic Brunisols (Soil Classification Working Group, 1998) developed on 1–50 m deep sand/silt glacial deposits overlying limestone bedrock. In 2010, this section of the Highway 401 had a daily traffic density (annual average) of 41,100 cars (Ministry of Transportation, 2010). A transect, perpendicular to the highway and extending approximately 1.5 km from the highway was established during the summer of 2013. For ground vegetation measurements, eight 20 m^2 plots were established at 23 m, 33 m, 84 m, 86 m, 121 m, 168 m, 237 m and 1483 m from the highway where the overstory was dominated by *Acer saccharum* (sugar maple) and canopy cover was similar (Table 1). Soil samples, throughfall and passive samplers for air chemistry were collected at 5 (33 m, 121 m, 168 m, 237 m and 1483 m) of these plots, plus an additional plot that was established 470 m from the highway. Epiphytic lichen surveys and lichen collection for chemistry were conducted on mature trees randomly selected along this gradient.

2.2. Throughfall deposition

Throughfall was collected using three collectors at each of the 6 sampling sites. During the spring, summer and fall, collectors consisted of 2 L plastic bottles topped with funnels that were fitted with nylon mesh filters to screen debris. In the winter months, a 15-L lined plastic bucket was used to collect snowfall. All throughfall collectors were placed approximately 2.5 m above the ground beneath dominant sugar maple trees. Throughfall was collected every 3 weeks between September 2013 and September 2014 resulting in a total of 17 collections through the year. Throughfall samples were filtered to 0.45 μm and measured for chloride (Cl^-), nitrate (NO^-) and sulphate (SO_4^{2-}) using a Dionex ICS-1100, ammonium (NH_4^+) using a SEAL Continuous Flow AutoAnalyzer and base cations (calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+) and sodium (Na^+)) and metals (iron (Fe), aluminum (Al), copper (Cu), zinc (Zn), manganese (Mn) and lead (Pb)) using ICP-OES. Charge balance calculations were performed on all samples to ensure they fell within 15%. Additionally, duplicate samples and blanks were run with each batch of samples to ensure quality control.

2.3. Passive air sampling

Atmospheric concentrations of NO_2 were measured with Willems badge passive samplers as described in Zbieranowski and Aherne (2012). Samplers were attached to tree stems approximately 1.8 m above the ground. Two samplers were placed at each plot and were changed every three weeks between July 24, 2013 and July 14, 2014. During each exposure period unexposed travel blanks ($n = 3$) were sent to and from sample sites to ensure that sampler caps and resealable bags were effective at protecting the samplers from contamination between preparation, exposure and analysis. Following removal from field sites samplers were refrigerated at 4 °C until analysis. Nitrogen dioxide samplers were analyzed by removing the absorbent pad from each sampler and immersing it in 5 mL of B-Pure water in a glove box. The solution was then agitated slightly and 2.5 mL were added to 1 mL of B-Pure water, to which 1.5 mL of reagent (2 g of sulfanilamide, 5 g of tartaric acid, 0.05 g of EDTA, 0.05 g of N-1-naphtylethyleme-diammonium dichloride, 5 mL of acetone, and diluted to 500 mL with B-Pure water) was added (Zbieranowski and Aherne, 2012). Samples were analyzed using a UV-VIS spectrometer at a wavelength of 540 nm (Zbieranowski and Aherne, 2012). A standard curve was attained using solutions of known NO_2 concentrations. Passive air sampler results were converted to average air concentrations using the formula:

$$C = (Q \times Rt)/(A \times t)$$

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