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2020-07-01

Yli-Pelkonen , V , Viippola , V , Kotze , J & Setälä , H 2020 , ' Impacts of urban roadside
forest patches on NO2 concentrations ' , Atmospheric Environment , vol. 232 , 117584 . https://doi.org/10.1016/j.atm

http://hdl.handle.net/10138/343366
https://doi.org/10.1016/j.atmosenv.2020.117584

Impacts of urban roadside forest patches on NO_2 concentrations

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Accepted final draft version

Reference:

Yli-Pelkonen, V., Viippola, V., Kotze, D.J., Setälä, H. (2020) Impacts of urban roadside forest patches on NO_2 concentrations. Atmospheric Environment 232, 117584.

http://doi.org/10.1016/j.atmosenv.2020.117584

Abstract

Although it is commonly believed that trees can improve air quality, recent studies have shown that such pollution mitigation can be negligible – or that tree canopies can even increase pollutant concentrations near their sources compared to adjacent treeless areas. We explored the impacts of urban roadside forest patches on the concentrations of nitrogen dioxide in summer and winter in the Helsinki Metropolitan Area, Finland, and especially investigated if canopy cover can result in increased concentrations of NO_2 below the canopy. Our results, however, did not show significantly higher – or lower – NO_2 concentrations underneath tree canopies compared to levels above canopies. Neither did NO_2 levels at the below-canopy sampling height differ significantly between forest patches and adjacent open, treeless areas. The lack of a canopy effect may derive from the rather small size of the forest patches, and – compared to previous studies with similar design – divergent tree species composition forming a dense canopy structure. Our results corroborate previous studies that the potential ecosystem services offered by urban near-road forests are more likely due to benefits other than those related to the removal of air pollutants.

Keywords: air pollutants; nature-based solutions; nitrogen dioxide; roads; tree canopy; urban forest

1. Introduction

Air pollution continues to be among the largest urban environmental problems worldwide. In many urban areas, concentrations of, for instance, nitrogen dioxide (NO_2) that originates from road traffic, energy production and industry, exceed safe levels for people and ecosystems (EEA, 2016). Traffic-related combustion is often the main source of nitrogen oxides ($NO_x = NO + NO_2$) of which the majority is emitted primarily as NO that is rapidly oxidized by ozone (O_3) to nitrogen dioxide (Anttila et al., 2011). Ground-level O_3 is formed when volatile organic compounds (VOCs) react with NO_x under sunlight; thus the concentrations of O_3 are dependent on NO_x and VOC emissions (e.g. Calfapietra et al., 2013; Chameides et al., 1992). VOCs are emitted from anthropogenic sources (e.g. traffic and industry) and natural sources (biogenic VOCs from e.g. trees). If local VOC concentrations are high, O_3 concentrations are very sensitive to NO_x concentrations. Furthermore, when NO_x concentrations are low, NO_x is a precursor to ozone formation, but at higher concentrations, NO_x catalyzes ozone destruction, resulting in O_3 depletion (e.g. Rodes and Holland, 1981). These complex interactions thus, in part, determine eventual NO_2 concentrations in urban environments. Elevated NO_2 levels may cause infections and respiratory symptoms in children and persons with asthma, in particular, as well as allergic and atopic symptoms (Krämer et al., 2000; Kampa and Castanas, 2008).

The influence of vegetation on urban air pollutant levels, and especially the potential of urban green to purify polluted urban air, has in recent years raised a considerable amount of research interests worldwide. Although the primary action for improving the quality of air ought to be the cutting of emissions (Duncan et al., 2016), it has been proposed – based on laboratory (e.g. Chaparro-Suarez et al., 2011; Hu et al., 2016) and modeling studies (e.g. Hirabayashi et al., 2012; Selmi et al., 2016) – that especially trees in urbanized settings capture air pollutants. For instance, absorption of, e.g. NO_2 into plant leaves (Rondón & Granat, 1994; Takahashi et al., 2005; Chaparro-Suarez et al., 2011) should improve urban air quality and provide a valuable ecosystem service or nature-based solution to the air pollution problem.

On the other hand, recent field studies have shown variable and often contradictory results on plant purification effects, the efficacy depending, e.g. on the studied air pollutant, climatic conditions and vegetation type and structure (e.g. Yin et al., 2011; Pataki et al., 2013; Setälä et al., 2013; Brantley et al., 2014; Fantozzi et al., 2015; Irga et al., 2015; Tong et al., 2015; Xing and Brimblecombe, 2019). Interestingly, recent studies by our research team and others have shown that concentrations of, for instance NO_2 (Harris and Manning, 2010; Yli-Pelkonen et al., 2017c; Viippola et al., 2018) in near-road environments and polycyclic aromatic hydrocarbons (PAHs) in near-road and park environments (Viippola et al., 2016; Yli-Pelkonen et al., 2018), can actually be higher under tree canopies than in open areas without trees. Such "negative" vegetation effects on local air quality can be considered an ecosystem disservice (Escobedo et al., 2011).

Reasons for the high concentration of pollution under tree canopies are not clear but we have suggested that they are likely due to polluted air being "trapped" under tree canopies due to reduced ventilation (Viippola et al., 2016; Yli-Pelkonen et al., 2017c; Viippola et al., 2018). Although trees can absorb NO_2 from the ambient air to some extent (Rondón & Granat, 1994; Chaparro-Suarez et al., 2011), we have suggested that the "trapping effect" may be high enough to mask uptake so that the net outcome is worse – or at least not better – air quality within tree canopies than in adjacent areas without trees (Yli-Pelkonen et al., 2017a, c; Viippola et al., 2018).

There are several mechanisms that can affect the dispersion of NO_2 from the pollution source, such as road traffic. NO_2 concentrations typically decline rather rapidly when moving further downwind from a road, however the decay curve is also dependent on traffic volume (Viippola et al., 2018; Yli-Pelkonen et al., 2017c; Xing and Brimblecombe, 2019). In near-road environments, solid barriers, such as noise walls or buildings, and semi-porous barriers, such as greenbelts or forests, can obstruct air movement and thus both horizontal and vertical pollutant transport and dispersion downwind from the pollution source (Abihijith et al., 2017; Baldauf, 2017). It has been suggested that air pollutant concentrations can increase between roads and solid barriers (Baldauf et al., 2008; Hagler et al., 2011) or greenbelts (Al-Dabbous and Kumar, 2014, Tong et al., 2016; Yli-Pelkonen et al., 2017c) due to the formation of a recirculation zone of air in front of a barrier. As shown by Yli-Pelkonen et al. (2017c), increasing density of trees can result in higher NO_2 concentrations between the road and the front edge of a forest.

Furthermore, vertical transport of NO_2 can be affected by barriers, such as greenbelts, due to increased turbulence in front of the barrier that can, depending on barrier height, elevate the air pollutant plume higher and over the barrier (Baldauf, 2017; Ghamesian et al., 2017). Empirical field studies focusing on the vertical distribution of NO_2 concentrations spanning over 50 m downwind from the pollution source are practically non-existent, but computational fluid dynamic simulations predicting general contours of pollutant concentrations for scenarios with or without barriers exist (Ghasemian et al., 2017).

Building on our aforementioned research on the topic, our current study aimed to further explore the canopy trapping effect by studying NO_2 levels in roadside forest patches under summer (dense foliage) and winter (thin, leafless, pervious canopy) conditions in Finland – and this time at two heights: below and above the canopy. Based on our earlier studies, our hypothesis is that

(1) the levels of NO_2 below tree canopies are higher than (a) those above canopies, and (b) those measured at the same height from the ground and distance from the pollution source in adjacent treeless areas. Our expectation is that (2) the levels of NO_2 above tree canopies are indifferent compared to concentrations at the same height and distance from the pollutions source in treeless areas. Furthermore, we expect that (3) the potential trapping effect is greater during summer (with leaves) than winter (without leaves).

2. Materials and methods

2.1. Study design and sampling

We studied the concentrations of NO_2 by measurements with diffusive passive samplers on the northern side of east-west oriented roads with large traffic volumes in the Helsinki Metropolitan Area (60°10′15″N, 24°56′15″E), southern Finland (Table 1). Altogether nine sampling sites were set up in three cities: five in Helsinki, three in Vantaa and one in Espoo (Fig. 1 and Appendix A). Each site included one forest patch area and one open area without trees. No biasing roads or intersections were nearby. The measurements were done 20 June – 28 July, 2017 (summer, full leaf-cover) and 7 November – 15 December, 2017 (winter, leafless period). The samplers were downwind from the roads for most of the measurement time (Fig. 2).

The treeless open areas were grasslands or meadows with short vegetation and either completely permeable or partly impermeable to water. The forest patches were dominated by broad-leaf young or semi-mature deciduous trees typical to southern Finland (*Acer platanoides*, *Alnus incana*, *Betula pubescens*, *Salix* spp., *Sorbus aucuparia*) and planted, non-native tree species (*Crataegus* spp., *Sorbus ulleungensis*, *Syringa vulgaris*).

The NO_2 samplers were placed underneath rain shields that were mounted to poles, aluminum pipes or trees at two heights (below and above the canopy). The collectors below the canopy were at 2.0 m height while the ones above the canopy situated 3.8 – 5.9 m (mean = 4.8 m) above ground, being 0.5 – 1.0 m above the canopy. Trees in the forest patch habitats were rather low allowing us to place the passive samplers above the canopy. In the treeless open habitats, the NO_2 collectors we placed at equivalent heights as in the forest patches.

At each site the distance of the samplers from the road was the same in both habitat types (forest patch or open). Due to varying forest patch sizes and locations in relation to the road, the distance of the samplers from the road varied among sites (see Table 1 and Appendix A for site details). The distance of the collectors from the forest patch front edge ranged from 5 to 15 m (mean = 8.9 m) (Table 1).

The passive, diffusive NO_2 collectors are developed by the Swedish Environmental Institute IVL and, in this study, were manufactured and analyzed by Metropolilab, Finland. The diffusive-collection method and laboratory analysis of NO_2 are described in detail in, e.g. Yli-Pelkonen et al. (2018). We used single sets of NO_2 samplers as these IVL-type samplers have proven very reliable and correlate strongly with continuous NO_2 monitors (Ferm and Rodhe, 1997; Ayers et al., 1998; Krupa and Legge, 2000; Loukkola et al., 2004; Kaski et al., 2016; Klingberg et al., 2017). For instance, Klingberg et al. (2017) measured NO_2 using IVL-type passive samplers in parallel to continuously monitoring NO_2 instrument (Tecan CLD 700 AL) and found a strong correlation ($r = 0.96$) between the two methods. Air temperature, which can influence NO_2 absorption (Loukkola et al., 2004), was monitored using Tinytag TG-4080 thermometers (accuracy: 0.01 °C) manufactured by Gemini Data Loggers Ltd.

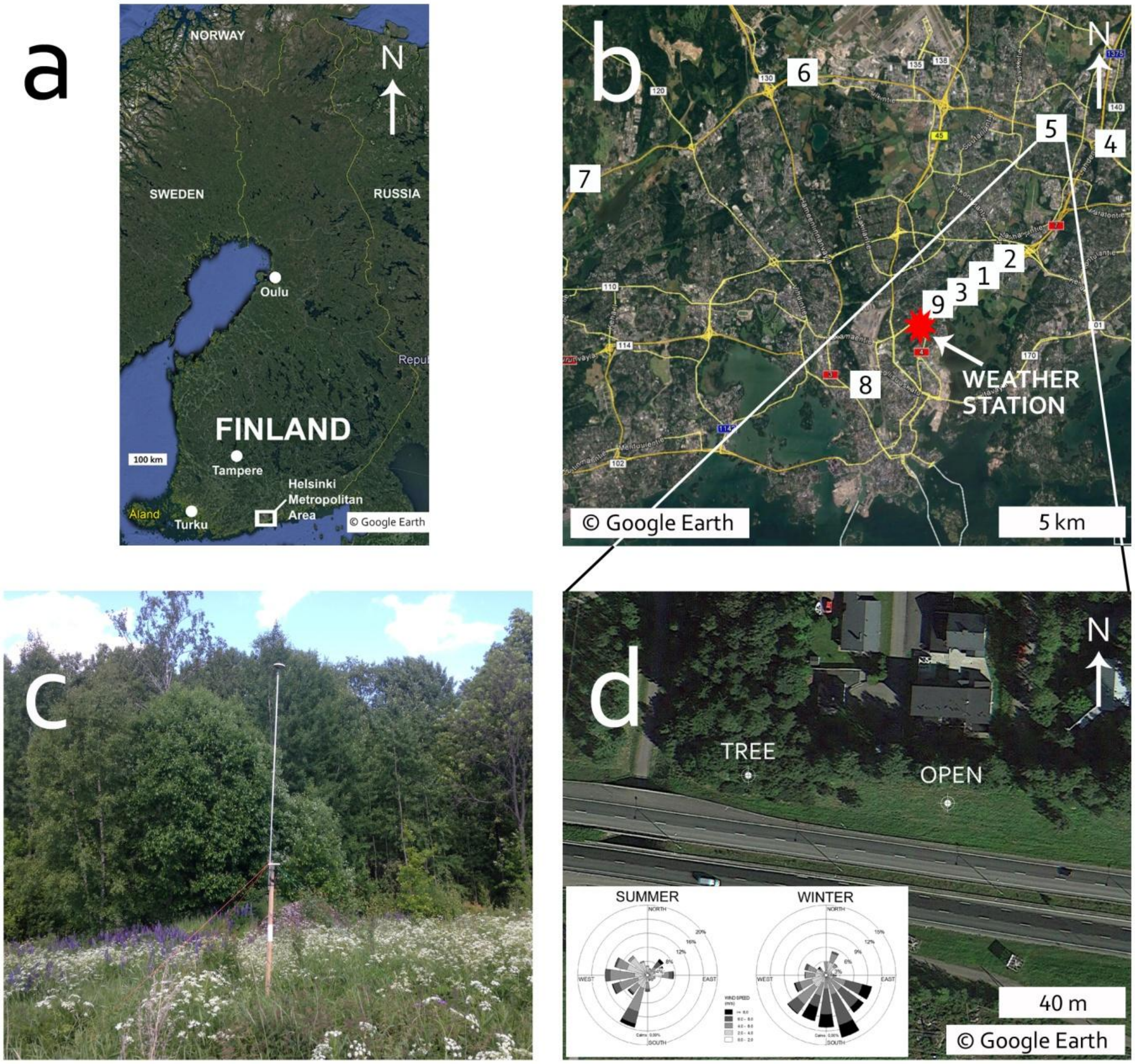

Fig. 1. (a) Helsinki Metropolitan Area (HMA) in Finland, (b) the nine study sites in the HMA and the location of the Kumpula weather station, (c) NO_2 sampler setup in an open area (two heights), and (d) one of the sites (site 5) as an example. Wind roses in (d) depict the wind patterns at the site during summer (left) and winter (right) measuring periods (see Fig. 2 for larger wind rose images). NO_2 was sampled within a forest patch ("TREE") at two heights: below and above the canopy – and at equivalent heights in an adjacent open area without trees ("OPEN") at each site. At site 5 the below- and above-canopy sampling heights were 2.0 m and 5.9 m above ground, respectively. See Appendix A for aerial images of each site.

Mean daily temperatures during the summer measurement period were 14.6 – 16.0 °C (mean = 15.1 °C) below the tree canopy, 15.6 – 16.7 °C (mean = 16.2 °C) above the tree canopy, and 15.5 – 16.7 °C (mean = 16.1 °C) at the lower measuring height in the open areas. Mean daily temperatures were significantly lower (a) below than above the canopy (6.4% lower, $p < 0.001$, $n = 9$) and (b) below the canopies compared to the lower measuring height in the treeless areas (6.0% lower, $p < 0.001$, $n = 9$) (paired samples *t*-test used). Mean daily temperatures did not differ between sampling heights or the forest and open areas during the winter measurement period, the mean being 2.1 °C. In the region, the measured temperatures are typical for June - July and November - December (FMI, 2017).

Table 1. Sampling and forest patch edge distances from the road (m), sampling heights from the ground (m), width of the forest patch (perpendicular to the road) (m), length of the forest patch (parallel to the road) (m), forest patch area (m^2), distance between sampling points (forest patch or open) (m), traffic flow of all vehicles (annual average of motor vehicles day^{-1}) and of heavy vehicles (only buses and trucks) (FTA, 2018) at the study sites. See Appendix A for aerial images of each site, including schematic dimensions of each site.

	Distance from the road edge			Sampling height							
Site nr.	Forest patch front edge	Sample point	Forest patch back edge	Below canopy	Above canopy	Forest patch width	Forest patch length	Forest patch area	Distance between forest & open sample point	Traffic flow, all vehicles	Traffic flow, heavy vehicles
1	5	11	17	2.0	4.1	12	85	950	35	48,000	3,200
2	6	12	16	2.0	4.0	10	88	915	79	48,000	3,200
3	6	16	17	2.0	3.8	11	113	1,050	82	48,000	3,200
4	32	42	74	2.0	4.7	42	105	3,200	114	43,000	3,900
5	7	15	75	2.0	5.9	68	60	2,990	55	57,000	4,800
6	21	26	33	2.0	4.8	12	26	530	45	76,000	7,100
7	10	25	43	2.0	5.9	33	100	1,673	41	45,000	4,000
8	5	17	24	2.0	5.6	19	17	234	37	28,000	1,500
9	3	11	212	2.0	4.8	209	263	35,000	58	48,000	3,100
mean	10.6	19.4	56.8	2.0	4.8	46.2	95.2	5,171	60.7	49,000	3,800

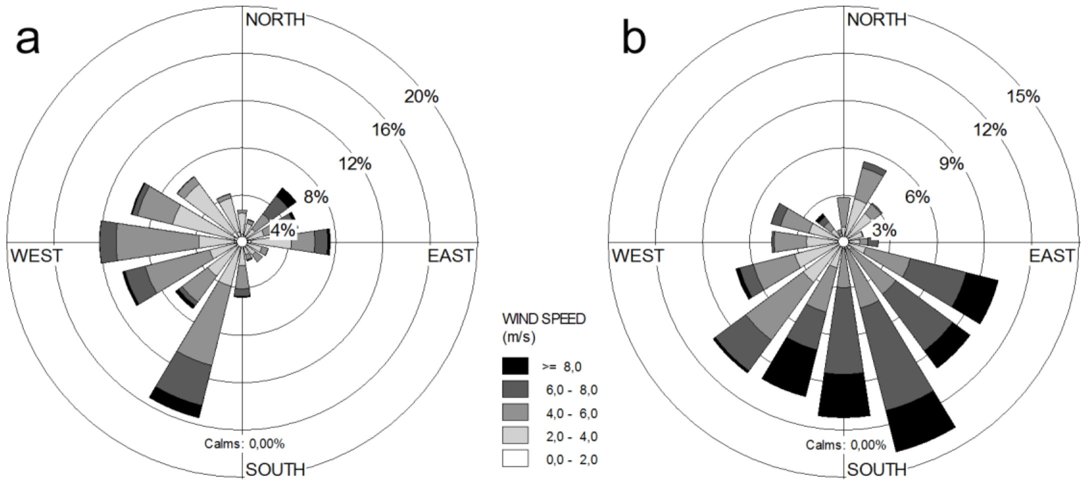

Fig. 2. Wind patterns in Helsinki (a) 20 June – 28 July, 2017 and (b) 7 November – 15 December, 2017 (FMI weather measurement station Kumpula).

2.2. Data analysis

We used general linear mixed models to test the effects of habitat type (forest and open), height (two separate heights: below the canopy, where the height was always 2 m from the ground and above the canopy, where the mean above-canopy measuring height 4.8 m was used) and season on NO_2 concentrations. Two types of analyses were performed using the *nlme* library in the statistical package R (R Core Team, 2017). First (model A), we analysed the effects of habitat type, height and season, and their two- and three-way interactions on NO_2 concentrations. Interactions were removed if they turned out to have little effect ($p > 0.1$). Second (model B), summer and winter data were analyzed separately, including habitat type and height, and their interaction into the model. Again, model selection removed interactions where appropriate. The reason for performing separate summer and winter models is that NO_2 concentrations are known to be higher during winter than summer in Finnish cities (Kaski et al., 2016) because of cold and calm circumstances when mixing and dilution of polluted air is less efficient. Indeed, when season was included in the models (model A), it produced a highly-significant signal in the results, which may mask potential effects of other predictors in the models. Due to the nature of the study design (with paired forest and open habitat sites and multiple samples per site, i.e., at two heights), site was included as a random term. In all instances, NO_2 concentrations were ln-transformed to satisfy assumptions of normality.

Since sampling points above the canopy varied between 3.8 – 5.9 m (mean = 4.8 m) from the ground and this variation could not be taken into account in the general linear mixed models, we explored the relationship between height above the canopy with NO_2 concentrations, using simple regression analysis. This analysis excluded all sampling points below the canopy (which were all at 2 m height).

3. Results

Mean NO_2 concentrations did not differ between forest and open areas or between the two sampling heights – representing the height levels below and above the canopy. The concentrations of NO_2 were significantly higher in winter than in summer (Model A: Table 2, Fig. 3). When data from two seasons (summer and winter) were analyzed separately (model B), mean NO_2 levels did not differ between forest and open areas, or between the two sampling heights in winter. However, sampling height did have an effect in the summer model, with significantly higher NO_2 concentrations at the lower sampling height at both habitat types (Table 2, Fig. 3).

NO_2 concentrations correlated negatively with sampling height above the canopy in summer in the forest patches ($r = -0.77$, $p = 0.016$) and in winter in the open areas ($r = -0.76$, $p = 0.018$), and marginally so in summer in the open areas ($r = -0.66$, $p = 0.055$) and in winter in the forest patches ($r = -0.71$, $p = 0.076$) (Fig. 3). Further exploration of the data shows no obvious relation between NO_2 concentrations (at various heights, in the forest or open areas, in winter or summer) and various traffic- and environmental-related variables, including traffic volume, heavy traffic volume, distances between the road and the forest patch and the sampling point, distance between the forest patch edge and the sampling point and forest patch size ($p > 0.050$ in all cases).

Table 2. Linear mixed effects model results. The following effects were tested: A) habitat type (forest and open), sampling height (below and above the canopy), season and their two- and three-way interactions on NO_2 concentration in the air. Model selection resulted in the removal of the three-way interaction and two-way interactions, B) habitat type and sampling height (and their interaction) on NO_2 concentration per season (summer and winter data analysed separately). Two-way interactions were removed after model selection. NO_2 was ln-transformed to satisfy the assumptions of normality. Statistically significant effects are in bold.

	Coefficient	SE	p
A: Three-way interaction model			
Intercept	2.759	0.075	< 0.001
Habitat (open)	-0.002	0.016	0.893
Sampling height (below)	0.025	0.016	0.121
Season (winter)	**0.510**	**0.016**	**< 0.001**
B: Summer model			
Intercept	2.753	0.079	< 0.001
Habitat (open)	-0.006	0.018	0.757
Sampling height (below)	**0.041**	**0.018**	**0.031**
B: Winter model			
Intercept	3.275	0.074	< 0.001
Habitat (open)	0.001	0.014	0.919
Sampling height (below)	0.008	0.014	0.544

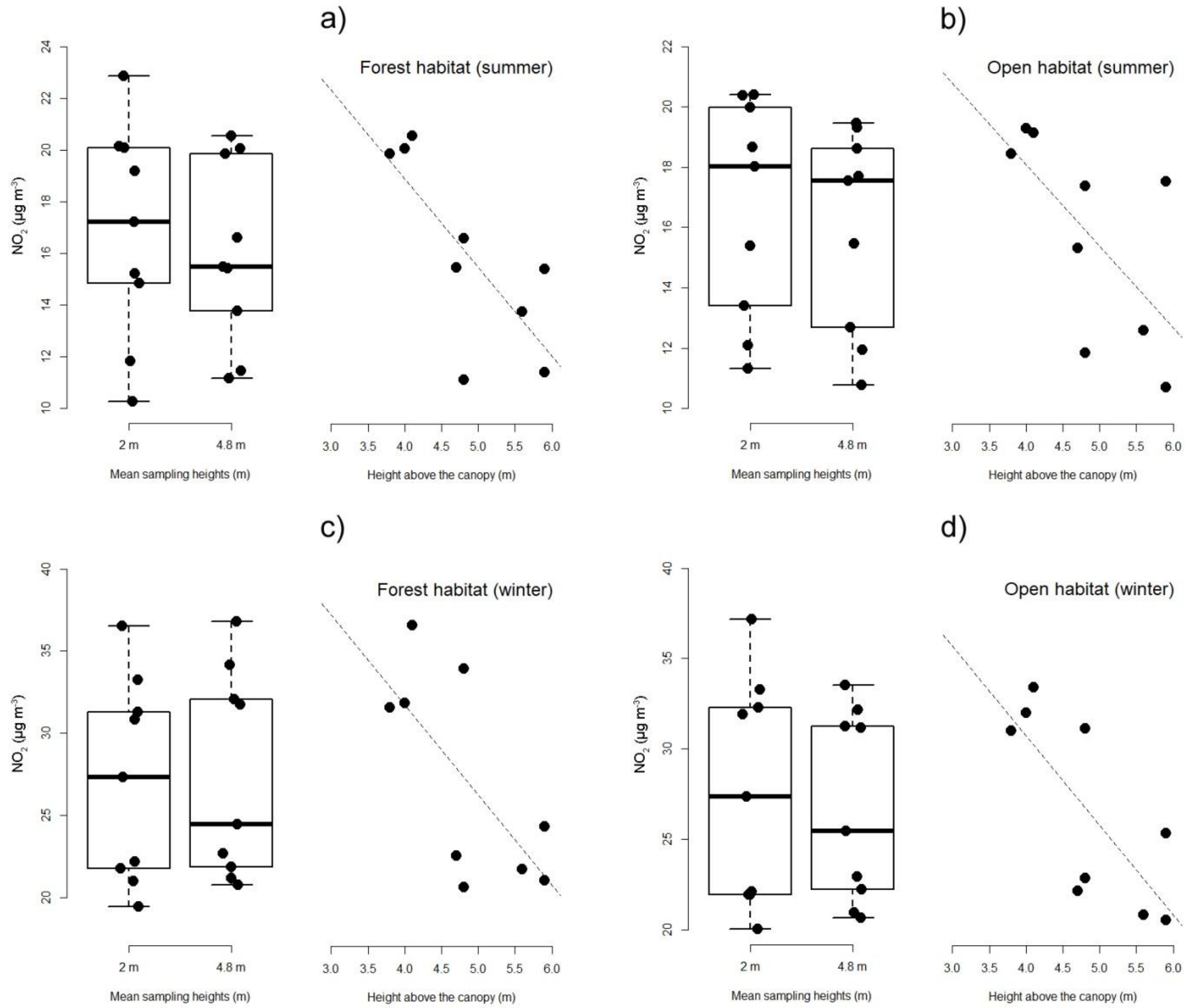

Figure 3. Concentrations of NO_2 in the summer (a) at heights below and above the canopy in the forest habitat, and (b) at equivalent heights in the open habitat, and in the winter (c) at heights below and above the canopy in the forest habitat, and (d) at equivalent heights in the open habitat (n = 9 sites). The lower measuring height (below-canopy level) was always 2.0 m (1st boxplot), while the mean higher measuring height (above-canopy level) was 4.8 m (2nd boxplot), with a range between 3.8 - 5.9 m (3rd plot, see main text for correlations). The individual data points are scattered in the boxplots (which display median values, quartiles and whiskers), using the *scripchart* function in R.

4. Discussion and conclusions

Contradicting our first hypothesis, concentrations of traffic-derived NO_2 were not statistically significantly higher below than above the tree canopies of mainly broadleaved forest patches. Neither did NO_2 levels at the lower sampling height differ significantly between forest patches and the adjacent open, treeless areas. However, that NO_2 concentrations above tree canopies (at a mean height of about 5 m) did not differ significantly between forested and open areas supports our second hypothesis, giving no indication that trees affect NO_2 concentrations right above the canopy either.

Although summertime NO_2 concentrations were slightly and significantly higher at the measurement height below the canopy than above it, this was the case in both forest and open areas and is likely due to the general dilution of NO_2 concentrations with increasing height from the tailpipe level of vehicles and road/ground surface (Restrepo et al., 2004). Likely for the same reason, NO_2 concentrations above the canopy decreased significantly as the sampling height increased, although only marginally so in summer in the open areas and in winter in the forest patches. Thus, we found little evidence of the "trapping effect" on NO_2 that we have in our previous studies suggested to result in clearly elevated pollutant concentrations in relation to adjacent areas without tree-cover (e.g. Yli-Pelkonen et al., 2017c; Viippola et al., 2018). That the vegetation-related NO_2 concentration patterns were equal in summer and winter further argues against the existence of a significant trapping effect in the current study, thereby refuting our third hypothesis. However, as we did not detect lower NO_2 concentrations below the canopies compared to open habitats, our current study corroborates our earlier results on the negligible impact of near-road forests in reducing NO_2 concentrations (Setälä et al., 2013; Yli-Pelkonen et al., 2017b, c; Viippola et al., 2018) as well as conclusions of the Air Quality Expert Group

(2018) according the which vegetation does not efficiently reduce NO_2. It is noteworthy that several previous studies by other researchers have reported reduced NO_2 levels by forests in near-road environments (e.g. Grundström and Pleijel, 2014; Fantozzi et al., 2015; Klingberg et al., 2017).

A possible explanation for why we did not detect a "trapping effect" in the current study compared to our previous studies, which showed elevated pollution levels inside the canopy, is that forest areas or patches were generally larger and trees taller in these studies. It is possible that such forest patches with larger volume below/within the canopy are more prone to "trap“ gaseous air pollutants such as NO_2 due to reduced ventilation within the canopy (Yli-Pelkonen et al., 2017c; Viippola et al., 2018).

It is possible that the denser and smaller forest patches in the current study were less-porous for polluted air currents to penetrate or flow through them (Wuyts et al., 2008) and thus the polluted air flowing from the road did not concentrate below canopies to such an extent that NO_2 levels would have been higher than at the other measuring points of the study set-up. Moreover, it also appears that tree species in the current study were not capable of absorbing NO_2 to a quantifiable extent. It cannot be ruled out, however, that the tree species here – somewhat different than in our previous studies – actually absorbed NO_2 to such an extent that the potential "trapping effect" was compensated for by the absorption of NO_2 and thus no difference was detected in NO_2 levels below the canopy in relation to the comparison points.

Fowler (2002) and Harris and Manning (2010) suggested that NO emissions from soil could be partly responsible of the increased NO_2 concentrations below tree canopies due to NO-producing soil organisms and because NO in the air is rapidly oxidized by O_3 to form NO_2. In principal, this could be a factor resulting also in the perceived inability of the tree canopy to reduce NO_2 levels, but in our study this would be an unlikely explanation as at all sites, except one, the tree-covered and open areas were equally covered with permeable soil and the pattern was the same both in summer and in winter, when soil is frozen and the ground is covered by snow.

Measured NO_2 concentrations in the current study (ranging between 10-23 μg m^{-3} in summer and 19-37 μg m^{-3} in winter) are generally in line with NO_2 concentrations measured in 2017 by the Helsinki Region Environmental Services Authority (Malkki et al., 2018), whose annual mean NO_2 concentrations ranged from 4 μg m^{-3} in the outskirts of the urban area to 33 μg m^{-3} in the urban core. Moreover, NO_2 concentrations in the current study follow the same levels as in our previous studies in the Helsinki Metropolitan Area where NO_2 concentrations ranged from 11 to 28 μg m^{-3} in summer and from 12 to 43 μg m^{-3} in winter. However, NO_2 concentrations in the region largely depend on traffic volume at the sampling sites and on distance from the road. It is typical in the Helsinki area that the main NO_2 source close to high-traffic roads is indeed road traffic (Ilmatieteen laitos, 2016; Yli-Pelkonen et al., 2017b, c) with "background" concentrations having a minor influence. NO_2 levels are rather low in the Helsinki Metropolitan Area and such background concentrations are usually detectable only about 100 m and more from roads with similar traffic volumes as in our study. Based on this, and also that our samplers were downwind from the roads for most of the measurement times, it is likely that NO_2 pollution at each of our study sites mainly derived from road traffic.

We showed that sampling distance – which varied between 11 and 42 m – from these high-traffic roads did not affect NO_2 concentrations significantly. Typically, NO_2 concentrations right by the road (1-3 m from the road edge) are the highest (e.g. Fantozzi et al., 2015; Gadsdon and Power, 2009), but then decrease markedly already at 10-20 m from the road edge, after which the decay curve becomes rather gentle (e.g. Xing and Brimblecombe, 2019; Yli-Pelkonen et al., 2017c). Consequently, it is unlikely that horizontal or vertical drop-off rates of NO_2 concentrations from road traffic created a significant bias in our study within the distance of 11-42 m from the road edge.

In our current study the measured NO_2 concentrations did not exceed the annual human health limit of 40 µg m^{-3} in Finland, likely because our samplers were located 11-42 m from the roads and not right by the roads. Moreover, according to the Helsinki Region Environmental Services Authority, the mean annual NO_2 concentrations have been decreasing slightly during recent years in the region (Malkki et al., 2018). Mean O_3 concentrations in similar near-road environments in the Helsinki Metropolitan Area typically range between 32-45 µg m^{-3} (Yli-Pelkonen et al., 2017b), thus O_3 likely contributed to NO_2 formation at the sites. However, as O_3 concentrations were not measured in this study, we cannot compare NO_2 concentrations directly with those of O_3.

To summarize, we showed that NO_2 concentrations are not significantly increased or reduced below canopies of relatively small deciduous forest patches near busy roads, although both increases and reductions of gaseous pollutants have been demonstrated in previous studies by our research team and by others. Nevertheless, our current results corroborate our earlier studies according to which near-road forests do not improve air quality locally regarding NO_2. Thus, the results of our current study suggest that the benefits offered by forest patches in urban near-road environments are more likely other kinds of ecosystem services or nature-based solutions (e.g. Lindén et al., 2020; Silvennoinen et al., 2017; Viippola et al., 2018) than those related to the removal of nitrogen dioxide from the air. In the future, it would be fruitful to conduct similar kinds of studies in even more urbanised settings with higher NO_2 concentrations, with different plant types and configurations, in a milieu with buildings and other urban infrastructure that can influence air flow patterns, as well as with other critical air pollutants. Furthermore, for a more complete seasonal understanding of the influence of tree-cover on NO_2 concentrations, sampling throughout the year is needed.

Acknowledgements

This study was funded by the City of Helsinki, the Maj and Tor Nessling Foundation and the Helsinki Metropolitan Region Urban Research Program (EKO-HYÖTY project). We acknowledge Taru Hämäläinen and Ronja Jokirinne for field work assistance and the anonymous reviewers for their constructive feedback and suggestions to improve the paper.

Appendix A. Supplementary data

The following is the Supplementary data to this article:

<link to PDF file> (see the official publication)

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