Unseen Emissions: Health Risks Cut by Shifting from Gas to Electric Cooking in Real Homes

Report by Greenpeace Research Laboratories, Greenpeace CEE Croatia and Greenpeace Hungary

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Abstract

Indoor air quality is increasingly recognised as important for health. A large proportion of people's time is spent at home, and it is common to spend more time breathing indoor air than outdoor air. This makes houses an important setting for exposure to air pollution. Cooking is an important source of indoor air pollutants. Here observations of indoor air pollutants (Nitrogen dioxide and Volatile Organic Compounds) were made in 9 Croatian and 3 Hungarian homes before and after gas cooking appliances were replaced with electric equivalents, to investigate the contribution made by gas combustion to levels of indoor air pollutants.

Across all participating homes the average measured indoor concentrations of nitrogen dioxide (NO_2), benzene and xylene were higher in Phase 1 (gas stove use) than in Phase 2 (electric stove use) of the study. Although measured outdoor concentrations of all compounds also decreased between phases, absolute reductions indoors were greater than outdoors for NO_2 in 8 of the 12 households and for benzene in 9 of the 12 households.

Greater indoor than outdoor reductions in NO_2 and benzene concentrations across the majority of households between Phase 1 and Phase 2 strongly suggests that removal of the gas cooking stove contributed to improved indoor air quality. Model based estimates of the contribution of indoor and outdoor sources to indoor NO_2 and benzene concentrations support this conclusion.

Changes in the measured concentrations of VOCs apart from benzene were more complex, likely the result of emissions from other sources such as paints and cleaning agents, air fresheners and building materials, either within the households investigated or perhaps in adjacent or nearby households.

These results indicate that cooking fuel choice can impact exposure to nitrogen dioxide and benzene, both chemicals with known adverse health effects. The results contribute important policy relevant data relating to public health, energy supply and planning. Actions to reduce public health risks could include prohibiting the installation of gas stoves in new residential buildings, indoor air quality regulations for nitrogen dioxide and health relevant VOCs, and the provision of financial support to replace gas cooking equipment with electric alternatives. Priorities for financial support should be homes where vulnerable groups including children or those with existing health conditions are present, poorly ventilated or overcrowded homes and homes on low incomes.

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Introduction

Many people spend a large proportion of their time indoors, particularly at home. In Europe, on average, adults spend as much as 90% of their time indoors, while infants, the elderly and chronically ill may spend even more (WHO, 2013). Therefore, indoor air quality is a significant factor in relation to individuals' exposure to air pollutants. Poor indoor air quality is associated with adverse health effects (Public Health England, 2020).

The quality of indoor air is determined by the balance between sources of pollution indoors, ingress of pollution from outdoors and the loss of pollutants by deposition

and ventilation. In homes, common sources of air pollution include building materials, furnishings, the use of volatile products such as paints (during storage, application and drying), cleaning products and air fresheners, and combustion processes such as cooking and heating.

Fossil gas, sometimes known as natural gas, is a fossil fuel commonly used for domestic heating and cooking. Its principal component is methane but it also contains trace amounts of other chemicals including nitrogen and non-methane volatile organic compounds (VOCs).

Methane is a powerful greenhouse gas with a global warming potential approximately 80 times that of CO_2 over a 20-year timeframe (Intergovernmental Panel on Climate Change (IPCC), 2023). Methane escapes into the atmosphere across the fossil gas supply chain (Howarth, 2014), making its continued use incompatible with long-term climate goals.

Of the VOCs contained in fossil gas, several such as benzene, are hazardous air pollutants. These can be emitted into the homes of gas users by a combination of incomplete combustion, escape while lighting the appliance, and post-meter leaks (Lebel et al., 2022; Michanowicz et al., 2022). Benzene is classed as a known carcinogen by the International Agency for Research on Cancer. Laboratory and in-home measurements also suggest that gas cooking activities raise indoor VOC concentrations, including for benzene, to levels relevant for health (Kashtan et al., 2023). Indoor VOC concentrations can be highly variable due in part to the large number of emission sources and relatively small room volumes (Warburton et al., 2025). The food being cooked can also influence the VOCs emitted (Kumar et al., 2025).

Actively burning fossil gas for cooking emits nitrogen dioxide (NO_2) directly into the home of the user (Lebel et al., 2022; Moschandreas et al., 1986; Mullen et al., 2016; Singer et al., 2017). Gas cooking has been linked to increased and more variable indoor NO_2 concentrations, making the use of gas as a fuel in homes an exposure pathway for NO_2 (Brown et al., 2009; CLASP, 2023; Daouda et al., 2024; Dimitroulopoulou et al., 2001; EECA, 2025; Marbury et al., 1988; Melia et al., 1990; Spengler et al., 1983, 1979; TNO, 2023). For example, a study of 206 households across 7 European countries found that the average indoor sourced NO_2 concentrations in kitchens cooking on gas were significantly higher (at 21.1 $\mu g/m^3$) compared to kitchens using electricity (7.9 $\mu g/m^3$) (TNO, 2023). Making changes at home, including the removal of gas cooking appliances, can be effective at decreasing indoor NO_2 concentrations (Daouda et al., 2024; Paulin et al., 2014). NO_2 can worsen respiratory problems, trigger asthma attacks, and reduce lung function, especially in children and people with pre-existing conditions (Committee on the Medical Effects of Air Pollution, 2018).

As noted above, indoor air quality is determined by a large number of factors, including influence from outdoors, a wide range of possible indoor pollution sources, building design, ventilation and individual behaviour. Controlling these parameters to determine the effectiveness of interventions designed to improve indoor air quality presents a challenge to researchers. Variables can be controlled by designing studies that make use of purpose built kitchen laboratories, though a disadvantage of this approach is that the data gathered are not representative of real-world environments and human behaviour. An alternative, longitudinal, experiment design is to sample in homes that are undergoing a transition from gas to electric cooking. This approach has the advantage that the building and its occupants remain unchanged while the intervention is made, although factors such as seasonal effects relating to changing weather and cooking habits remain harder to control for.

This current study takes a longitudinal approach to investigate changes in indoor air quality that result from the replacement of gas cooking equipment with electric appliances. Previously researchers have used similar methods to investigate NO_2 concentrations in the United States (Daouda et al., 2024). Nonetheless, to our knowledge, the current study is the first of its kind to also assess changes in indoor VOC concentrations over such a transition.

The present study in Croatia and Hungary was coordinated by Greenpeace CEE Croatia and Greenpeace Hungary and the Greenpeace Research Laboratories. It investigates the extent to which transitioning from gas to electric-powered cooking can influence indoor air quality. The study made measurements of NO_2 and VOC concentrations in the indoor air of participants' home kitchens over two sequential phases of monitoring, one period before and one after they transitioned from gas to electric cooking.

Methods

Experiment design

Measurements of NO_2 and VOC concentrations were made in the kitchens of study participants over two 2-week periods. During the first period, Phase 1, the households' predominant cooking fuel was gas. Each household then independently replaced their gas cooker with an electric appliance. During the second period, Phase 2, the cooking fuel in all households was electricity. Phase 1 measurements were made in January and February 2025, while Phase 2 measurements were made in April and May 2025.

Participant recruitment

The study was conducted at participants' homes in Croatia and Hungary during 2025. Participants were recruited via direct email communication to existing Greenpeace CEE Croatia and Greenpeace Hungary supporters, and via social media advertisements. All households in Croatia that completed the two sampling phases were offered a single EUR 100 inconvenience allowance, while local regulations meant that it was not possible to offer such compensation in Hungary.

Eligibility criteria to be included as a participant in the study were that the household:

- predominantly used gas from a piped supply for cooking,
- independently planned to transition from gas to electric cooking within the project timeframe,
- used the cooker most days (minimum 4 days per week),
- did not use solid fuel burning including open fires,
- did not include smokers,
- would not use incense or candles during the study,
- was not on a heavily trafficked road,
- was not adjacent to an industrial site or petrol station, and that
- included participants that were not under 18 years of age.

14 households were included in Phase 1, 11 in Croatia and 3 in Hungary. Of these, two households from Croatia did not complete their transition to electric cooking during the study timeframe to be included in Phase 2. These households are not included in the analysis presented here.

Household questionnaires

Participating households were surveyed using structured questionnaires on three occasions: (i) before the start of the first monitoring period, (ii) after the first phase of sampling when the gas stove was present, and (iii) at the conclusion of the experiment following the second phase of sampling when the gas stove had been replaced with an electric stove. Information on the fuel type used for cooking, the weekly frequency of use of the stove or oven, if and how participants ventilated their kitchen during the study (cooker hood or/and opened windows), the approximate size of the kitchen (length, width, height, volume), and whether participants were aware of any other relevant indoor air pollution sources such as the use of solid fuels, use of incense, candles or smoking materials prior to the study, use of cleaning agents, glues or similar products or the installation of new furnishings. All participants used electric ovens throughout and so this is not considered further.

Nitrogen Dioxide

The sampling for NO_2 used passive Palmes type diffusion tubes (Palmes et al., 1976), with analysis by UV spectrophotometry. The tubes and analysis were provided by Gradko International Limited. This method provides quantitative estimates of the average concentration of NO_2 over the sampling period.

For each phase of the study, two triplicate sets of diffusion tubes were installed at each participating household. Three tubes were placed outdoors on the outside wall of the house as close to the kitchen as possible, and at a height of approximately 2-3 m. The other three tubes were installed indoors in participants' kitchens as close to 1 m from the cooker as possible, but not within 50 cm to minimise ingress of steam and particles from cooking, at a consistent sample height of approximately 1.5 m above ground level. All outdoor and indoor tubes were placed in areas where air can circulate. The diffusion tubes were deployed for approximately 14 days. During each phase of the study four travel blanks were included to assess sample integrity; these were transported with the sample tubes to and from the laboratory and to each participating household. All measured NO_2 concentrations were corrected by subtracting the highest blank value from the corresponding study phase.

VOCs

The sampling for VOCs was undertaken using Markes International EPA 325 thermal desorption (TD) tubes, which are packed with Carbopack X sorbent (Markes International, 2016). TD tubes for sample collection were freshly conditioned (380 C for 4 hours in a nitrogen flow of >50 ml/min) and their cleanliness verified using TD-GC-MS (see below) before being sealed with brass caps and wrapped in foil ready for dispatch from our laboratories. Tubes were deployed simultaneously with the NO_2 diffusion tubes, and using the same experimental design consisting of outdoor and indoor triplicates as described above. For VOC sampling, both field/travel blanks (tubes that were transported along with those used for sample collection, but which remained sealed at all times) and field/travel standards (preloaded in our laboratory with a mixed VOC standard - see below) were used to assess sample integrity (3 of each per country and sampling phase) and to control for cross contamination between tubes during transits. At the end of each sampling phase, TD tubes were retrieved from their indoor or outdoor locations and immediately resealed using the same brass caps, in order to preserve sample integrity during storage and transport. Tubes were not stored cold in order to avoid differential contraction of the different metal components, which could otherwise result in the caps loosening during storage and either ingress or egress of VOCs.

On return to the Greenpeace Research Laboratories, all VOC sorbent tubes were analysed by thermal desorption-gas chromatography-mass spectrometry (TD-GC-MS)

using a Markes International Centri sample extraction and enrichment platform (with a TD-50 stack), coupled to an Agilent gas chromatography - mass-spectrometry (GC-MS) system (Agilent Technologies).

Tubes were first dry purged for 1 minute at 50 C with a backflow of nitrogen at 50ml/min to remove any residual water vapour, before desorption at 320 C for 5 minutes on a forward flow of nitrogen (at the same flow rate). VOCs desorbed from each tube were initially collected on a peltier-cooled trap, which was subsequently purged at 25 C and 50 ml/min nitrogen for 1 minute before heating rapidly to 320 C (held for 5 minutes) in order to desorb the VOCs and direct them through a heated transfer line to an Agilent 7890B GC.

The GC inlet was operated in splitless mode, held at a temperature of 200 C and an initial pressure of 20 psi. VOCs then passed onto a 30m Restek Rxi-624 Sil capillary column (250 um internal diameter, 1.4 um phase thickness), held at an initial temperature of 40 C for 4 minutes following injection, using helium as an inert carrier gas. The oven temperature was then ramped at 15 C/min up to 240 C, held for 10 minutes, before ramping to 260 C at 50 C/min, held for 2 minutes, to ensure that there was no carry-over of VOC residues between samples, yielding a total GC run time of just under 30 minutes per sample. A constant flow of 1.6 ml/min helium was maintained throughout by ramping the column pressure from approximately 20psi to approximately 40 psi through the run.

Separated VOC compounds were identified and, in the case of the so-called ‘BTEX’ compounds (benzene, toluene, ethylbenzene, m/p-xylene and o-xylene), quantified using an Agilent 59877A MSD, operated in SCAN mode (3.55 scans/s), over a mass/charge (m/z) ratio range of 35-450. Data acquisition was performed using Agilent Enhanced Masshunter software, run in Chemstation emulator mode. Quantities of each BTEX compound which had diffused onto the TD tube sorbent during the sampling periods were determined through manual integration of chromatogram peaks and comparison of peak areas against those obtained for calibration standards.

Before analysing the samples, instrument calibrations were performed using a mixed VOC standard containing 16 individual VOC compounds, including the ‘BTEX’ species that were the subject of this investigation, along with a range of other non-chlorinated and chlorinated compounds to check for consistency in peak area and retention time across the majority of the GC temperature programme. To prepare the calibration standards, freshly conditioned TD tubes were loaded manually (using a Markes Calibration Standard Loading Rig with a gas flow of 50 ml/min nitrogen) with 1 µl of a serial dilution of mixed VOC standard, containing concentrations (for all but two of the 16 VOC compounds) ranging from 1 mg/l up to 300mg/l, providing a six point calibration in the range of 1 ng to 300 ng of each compound loaded on the TD

tubes. In the specific cases of the ortho- and para- isomers of xylene (o- and p-xylene), concentrations ranged from 2 mg/l up to 600 mg/l, yielding from 2 ng to 600 ng on each calibration tube. Deuterated toluene (toluene-d8) was included as an additional standard in all calibration checks as part of our routine system suitability and sensitivity checks, using a standard concentration of 30 mg/l, yielding 30 ng on each tube.

Due to the difficulty of separating the peaks for meta- and para- isomers of xylene (m- and p-xylene) by GC-MS, these two isomers were quantified together. Therefore, throughout this study, values for m- and p-xylene are reported as the sum of their concentrations, rather than as individual compounds.

In preparing the final calibration curves, we excluded the highest VOC loading (300 ng) as being beyond the linear range for most compounds, thereby reducing the final calibrations to 5 points rather than 6. We adopted linear regression models for the standard curves as they provided good fits to the selected calibration points ($R^2 \geq 0.99$, $p < 0.001$ for all target species).

Following instrument calibration, samples and blanks were run in successive order of household and location (indoor/outdoor). A freshly prepared calibration check standard, loaded with 30 ng of each VOC compound (including toluene-d8) except o- and p-xylene (60ng), was run every 6 to 18 samples, in order to check for and quantify sensitivity drift in the calibrations, allowing subsequent correction of the peak areas for the samples. A further 6 such check standards were analysed at the end of the sample set in each phase of the project to provide a check for, and an estimate of, reproducibility.

Where drift in the peak areas for these check standards was detected over time during a sequence of samples and blanks, peak areas were corrected by calculating a sample-reference peak area by interpolating between the closest two available check standards, assuming a linear drift in sensitivity over this time. We avoided interpolating between control points established on different days since machine drift was generally more pronounced overnight. When a day run did not start or end with a control, we therefore used the next or last available control point instead. We finally calculated the correction factor as the ratio between the sample-reference peak area and the standard-reference peak area, and applied this multiplicative factor to the respective measured peak area to obtain the drift-corrected peak area. Following this correction for sensitivity drift we converted the corrected peak areas into quantified tube contents using the established standard curves.

Field/travel standards were prepared by loading TD tubes (prior to dispatch) with freshly prepared mixed VOC standard to give 30 ng of each VOC compound on the tube, other than o- and p-xylene, for which 60 ng of each was pre-loaded, as well as

30 ng of toluene-d8. These tubes were sealed with brass caps in the same way as the sample and blank tubes, and like the blank tubes, remained sealed throughout the sampling phase.

Before being sent to the field along with the sample tubes and blanks, these field standards had been analysed by TD-GC-MS using the method set out above, recollecting a nominal 45% of the amounts of each VOC compound initially loaded on to the TD tubes back on to those same tubes from the cold trap. We used these field standards as quality controls by ensuring that, upon return from the field, they still contained approximately the nominal recollected quantities of the VOC compounds. All tubes were sealed with long-term storage caps for transport to and from their sampling locations.

For each sample we then calculated quantities of all BTEX compounds in all field/travel blanks. These fell below the method detection limit and therefore did not warrant further corrections to the quantities measured on the sample TD tubes themselves.

Ultimately, the quantities of each BTEX compound determined to have been adsorbed onto each tube were converted to average air concentrations during each sampling phase using published uptake rates and the exposure times for the respective sorbent tubes, according to the method described in Markes International application note (Markes International, 2016). We used the two-weeks uptake rates for benzene, toluene, ethylbenzene and p-xylene from (Brown, 2022) as reported in Markes application note AN001 (Markes International, 2023). Uptake rates for m-xylene and o-xylene are not specifically reported for the Carbopack X sorbent; therefore we used the p-xylene rate for all three isomers.

We hereunder report these average two-weeks indoor and outdoor concentrations as mean ± the standard error of the triplicate measurements.

It should be noted that, in addition to the BTEX compounds that were the subject of quantification in this study, diverse ranges of other VOC compounds had also been collected by the TD tubes during both Phase 1 and Phase 2 of this study. These are currently undergoing qualitative analysis in order to determine their identities and relative abundances (indicated by relative peak areas), and to determine if there are significant qualitative differences between households, between indoor and outdoor air across all households and between Phase 1 and Phase 2 in individual households. Results of those qualitative analyses, once completed, will be reported separately.

Air exchange model

To describe the flow of the pollutants and the exchanges between indoor and outdoor air we used a simple well-mixed mass balance model:

$$\frac{dC_{in}(t)}{dt} = \frac{S(t)}{V} + \lambda P C_{out}(t) - \lambda C_{in}(t) - k_d C_{in}(t) \quad (1)$$

with, for a given pollutant:
C_{in} the indoor concentration
C_{out} the outdoor concentration
S the rate of emission of indoor sources
V the indoor volume
λ the air change rate (ACR, number of air changes per hour)
k_d the deposition/degradation factor
Our two-week averages can be approximated by a steady state with both fixed indoor emission rate and outdoor concentration forcing:

$$s + \lambda C_{out} - (\lambda + k_d) C_{in} = 0 \quad (2)$$

with the four terms representing the contribution of indoor sources, imports from outdoors, exports to outdoors and deposition/degradation, respectively. To simplify we have set $s = S/V$ as the emission rate per unit of indoor volume.
The partitioning of the indoor concentration in its contributions due to indoor and outdoor sources $C_{in} = C_{in}^{(in)} + C_{in}^{(out)}$ (indoor-sourced and outdoor-sourced concentrations) is, by substitution in (2), given by:

$$C_{in}^{(out)} = \frac{\lambda}{\lambda + k_d} C_{out} \quad (3)$$

$$C_{in}^{(in)} = C_{in} - \frac{\lambda}{\lambda + k_d} C_{out} \quad (4)$$

We estimated the indoor-sourced concentrations for phases 1 and 2 respectively using (4), as has been done in similar studies (TNO, 2023).

We assumed that the deposition factor k_d was unchanged across the two phases of the experiment, but that the ventilation rates differed. The first phase of our study was conducted in January-February while the second phase took place in April-May, and we assume that milder temperatures made it likely that participants opened doors and windows more frequently, therefore increasing the ventilation rates of their kitchens.

No quantitative data on air change rate (λ) were collected as part of the study. Questionnaire results reveal that all participants used ventilation in their kitchen (open windows or extractor hoods that vent outdoors). Therefore we adopt typical air exchange rates from the literature for the respective periods. Field methods for estimating air exchange rate are challenging and previous studies suggest that, for gross assessments of ventilation over a long period, the status of open windows may provide sufficient information without the difficulty and expense of conducting AER measurements (Brown et al., 2009).

Values for λ in the published literature were reviewed and selected for use here. Factors influencing λ include climate and geography, building design and materials, and human behaviour. In a kitchen setting, and given that questionnaire responses collected from the study participants showed that all participants in both Phase 1 and Phase 2 actively increased ventilation in their kitchens, we consider that behaviour is likely to be the most important factor. We therefore select λ values reported by (Dimitroulopoulou et al., 2001), who investigated air exchange in kitchens in the United Kingdom and who report values for summer and winter conditions of 1.5 and 1 respectively. Applied separately to Phase 1 (winter) and Phase 2 (summer), this choice incorporates a representation of behavior change between colder and warmer seasons specific to a kitchen environment. The values are applied uniformly to all participants.

The NO_2 deposition rate (k_d = 1.05) is adopted from Yang et al. (2004) who averaged values from previous studies in 'western' countries (Nazaroff and Cass, 1986; Spicer et al., 1993, 1989; Traynor et al., 1982; Wade et al., 1975; Wilkes et al., 1996). There is very limited experimental evidence describing VOC deposition rates. Evidence from a museum with mechanical ventilation (Pagonis et al., 2019) and a chamber experiment designed to mimic a domestic setting (Singer et al., 2004) suggests that deposition via sorption to surfaces is of relatively little importance in controlling benzene concentrations, but can affect concentrations for toluene, ethylbenzene and o-xylene in situations where there are low air change rates. In the kitchen settings investigated here, sorption rates are anticipated to be much slower than air change rate such that VOCs are assumed to have been removed by ventilation before a significant amount of sorption to kitchen surfaces can occur. Additionally, over the 2-week sample period used here, VOC desorption from surfaces further reduces the effective deposition rate. We therefore set k_d to zero for VOC species in both Phase 1 and Phase 2. This simplification can be considered to be a conservative approach as it increases the estimated contribution of indoor sources counter to our hypothesis.

Comparative analysis of pollutants between Phase 1 and Phase 2

To ensure that we compared the two phases of the experiment in a similar seasonal context, we controlled for the effect of different ventilation rates on the dilution of pollutants in indoor air. The modeled indoor-sourced concentration depends only on the the emission, ventilation and deposition rates:

$$C_{in}^{(in)} = \frac{s}{\lambda + k_d} \quad (5)$$

Had the mean indoor emissions from Phase 2 instead taken place in the conditions of ventilation encountered in Phase 1, all other parameters being equal, it would from (5) have resulted in an indoor-sourced concentration of:

$$C_{in,2\to1}^{(in)} = \frac{s_2}{\lambda_1 + k_d} = \frac{\lambda_2 + k_d}{\lambda_1 + k_d} C_{in,2}^{(in)} \quad (6)$$

where $C_{in,2\to1}^{(in)}$ denotes the indoor-sourced concentration for Phase 2 adjusted to a common ventilation basis with Phase 1. Hereunder, we refer to this value as the *modelled* or *indoor-modelled* concentration.

Indoor-modelled concentrations were found to be non-normal for four out of the six pollutants (Shapiro-Wilk normality test), we applied for all pollutants the Wilcoxon signed-rank test to the paired values for $C_{in,1}^{(in)}$ and $C_{in,2\to1}^{(in)}$ to assess whether the differences were significant.

Results

Measured indoor concentrations of NO_2, benzene, toluene, ethylbenzene, m/p-xylene and o-xylene in Phase 1 and Phase 2 are summarised in Table 1 and Figure 1 and 2. During Phase 1, average NO_2 concentrations ranged between 13.2 and 77.6 µg/m^3, they were 3.3 and 16.9 µg/m^3 in Phase 2. During Phase 1 benzene concentrations ranged between 0.3 and 2.4 ppb, and were <0.1 and 0.5 ppb in Phase 2 (Table 1, Figure 1).

Of the 12 households completing the study, absolute indoor reductions were larger than outdoor reductions at eight households for NO_2, nine for benzene, eight for toluene, eight for ethylbenzene, seven for m/p-xylene and nine for o-xylene.

Model estimates of Phase 1 and Phase 2 indoor-modelled NO_2, benzene, toluene, ethylbenzene, m/p-xylene and o-xylene concentrations attempt to isolate the changes in indoor air quality due to indoor sources (Table 2, Figures 3, 4 and Appendix 1).

Average Phase 2 indoor-modelled model results are lower for NO_2, benzene and o-xylene, and higher for toluene, ethylbenzene and m/p-xylene.

Table 1. Measured mean, minimum and maximum Phase 1 and Phase 2 concentrations of NO_2, benzene, toluene, ethylbenzene, m/p-xylene and o-xylene for indoor and outdoor samples.

		NO_2		Benzene		Toluene		Ethylbenzene		m/p-xylene		o-xylene	
Phase	**Statistic**	**Out**	**In**	**Out**	**In**	**Out**	**In**	**Out**	**In**	**Out**	**In**	**Out**	**In**
1	Max	31.9	77.6	1.17	2.37	0.59	1.35	0.17	0.39	0.77	2.10	0.35	0.93
	Min	5.6	13.2	0.40	0.48	0.20	0.34	0.08	0.10	0.39	0.49	0.23	0.27
	Mean	20.8	28.3	0.68	0.86	0.40	0.91	0.12	0.21	0.51	1.12	0.27	0.48
2	Max	19.9	16.9	0.30	0.29	0.50	1.52	0.08	0.24	0.31	1.32	0.13	0.39
	Min	8.7	3.7	0.13	0.00	0.09	0.16	0.03	0.04	0.10	0.17	0.05	0.07
	Mean	13.5	9.4	0.19	0.12	0.27	0.72	0.06	0.13	0.20	0.65	0.08	0.20
Units		$\mu g/m^3$		ppb									

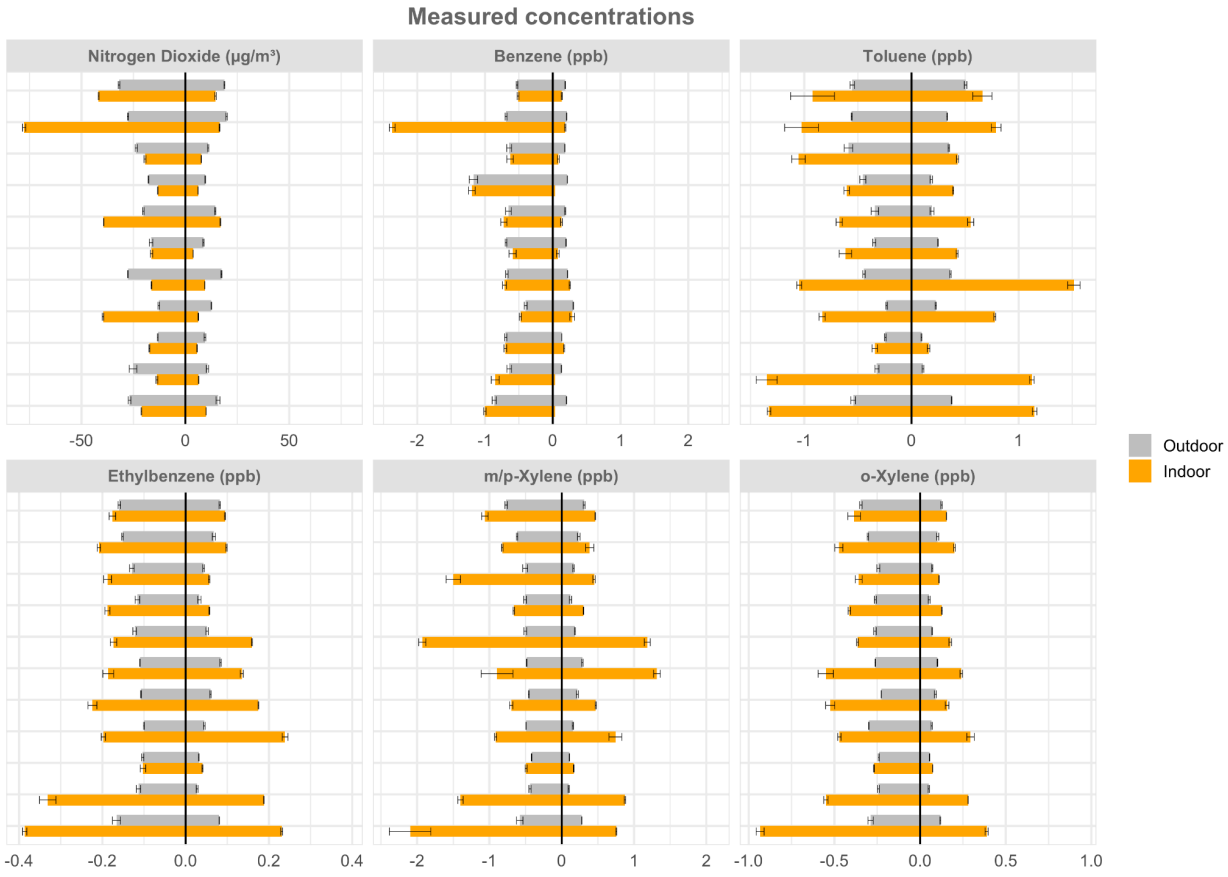

Figure 1. Measured (triplicate mean +/- standard error) Phase 1 (left) and Phase 2 (right) concentrations of NO_2 ($\mu g/m^3$), benzene, toluene, ethylbenzene, m/p-xylene and o-xylene (ppb) for indoor and outdoor samples at each household.

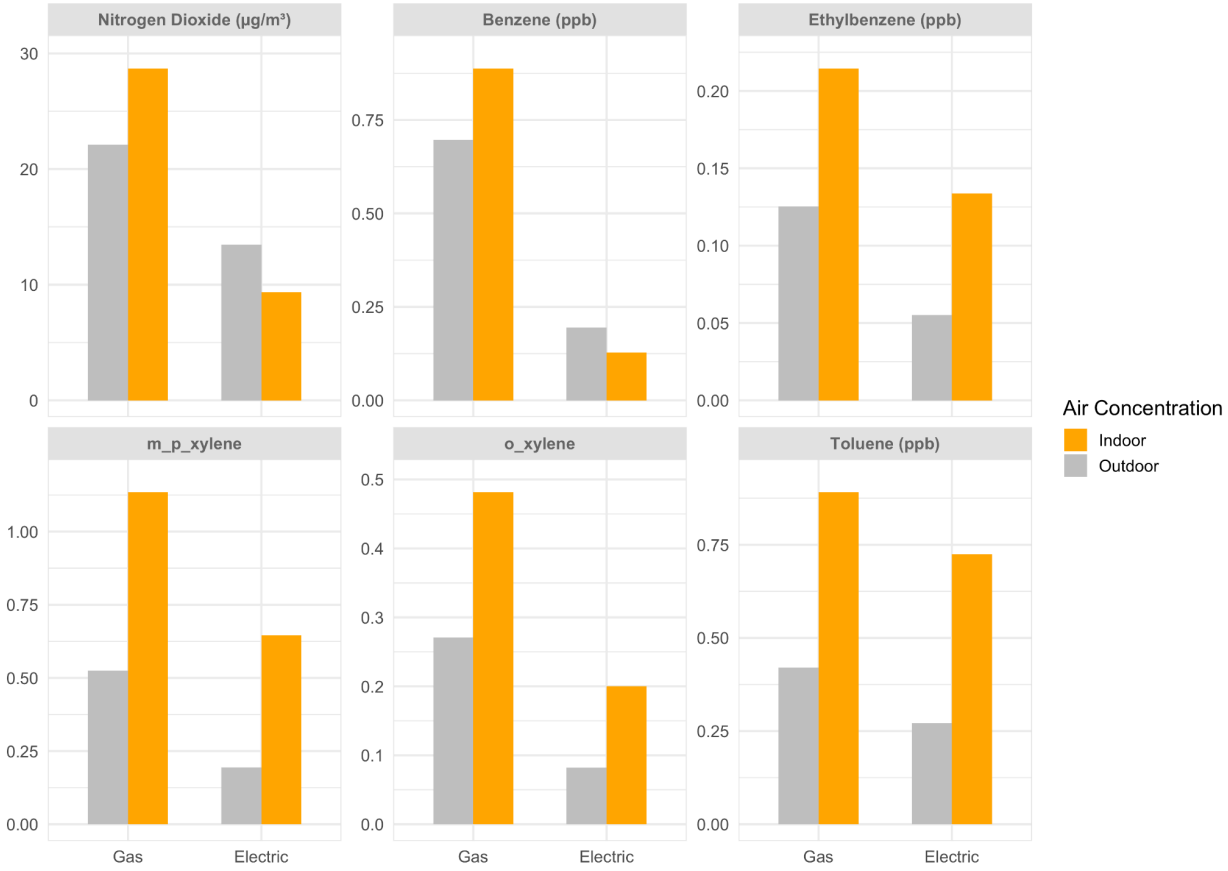

Figure 2. Average of measured phase 1 (gas) and phase 2 (electric) concentrations of NO_2 (µg/m^3), benzene, toluene, ethylbenzene, m/p-xylene and o-xylene (ppb) for indoor and outdoor samples.

The Wilcoxon non-parametric statistical test (confidence level 95%) was used to investigate the significance of differences between the Phase 1 and Phase 2 indoor-modelled results. This test compares a null hypothesis that there was no effect on air quality between Phase 1 and Phase 2 (i.e. that the median of the differences between Phase 1 and Phase 2 is zero) and an alternative hypothesis there is a change (i.e. that the median of the differences between Phase 1 and Phase 2 is not zero). In cases where the test's p value is < 0.05 the null hypothesis is rejected, meaning there is a statistically significant difference between Phase 1 and Phase 2.

In our results this condition is met for NO_2 and benzene. The results suggest that we can be 95% confident that the real change in indoor-source pollutant concentration was between the low and high confidence intervals shown in Table 2. Only NO_2 and benzene have consistently negative changes between Phase 1 and 2 at the 95% confidence level. The average reduction for NO_2 and benzene were 90% and 100% respectively.

Table 2. Indoor-model estimates of Phase 1 and Phase 2 NO_2, benzene, toluene, ethylbenzene, m/p-xylene and o-xylene, the Phase 1 to Phase 2 difference and results of a Wilcoxon test for the indoor samples.

	Mean			Wilcoxon		
Compound	**Phase 1**	**Phase 2**	**Difference**	**P**	**Low confidence interval**	**High confidence interval**
NO_2	17.91	1.79	-16.12	0.001	-31.01	-5.50
Benzene	0.19	0	-0.19	0.005	-0.84	-0.05
Ethylbenzene	0.09	0.12	0.03	0.240	-0.02	0.08
m/p-Xylene	0.61	0.68	0.07	0.365	-0.30	0.33
o-Xylene	0.21	0.18	-0.03	0.320	-0.12	0.04
Toluene	0.47	0.68	0.21	0.083	-0.06	0.49

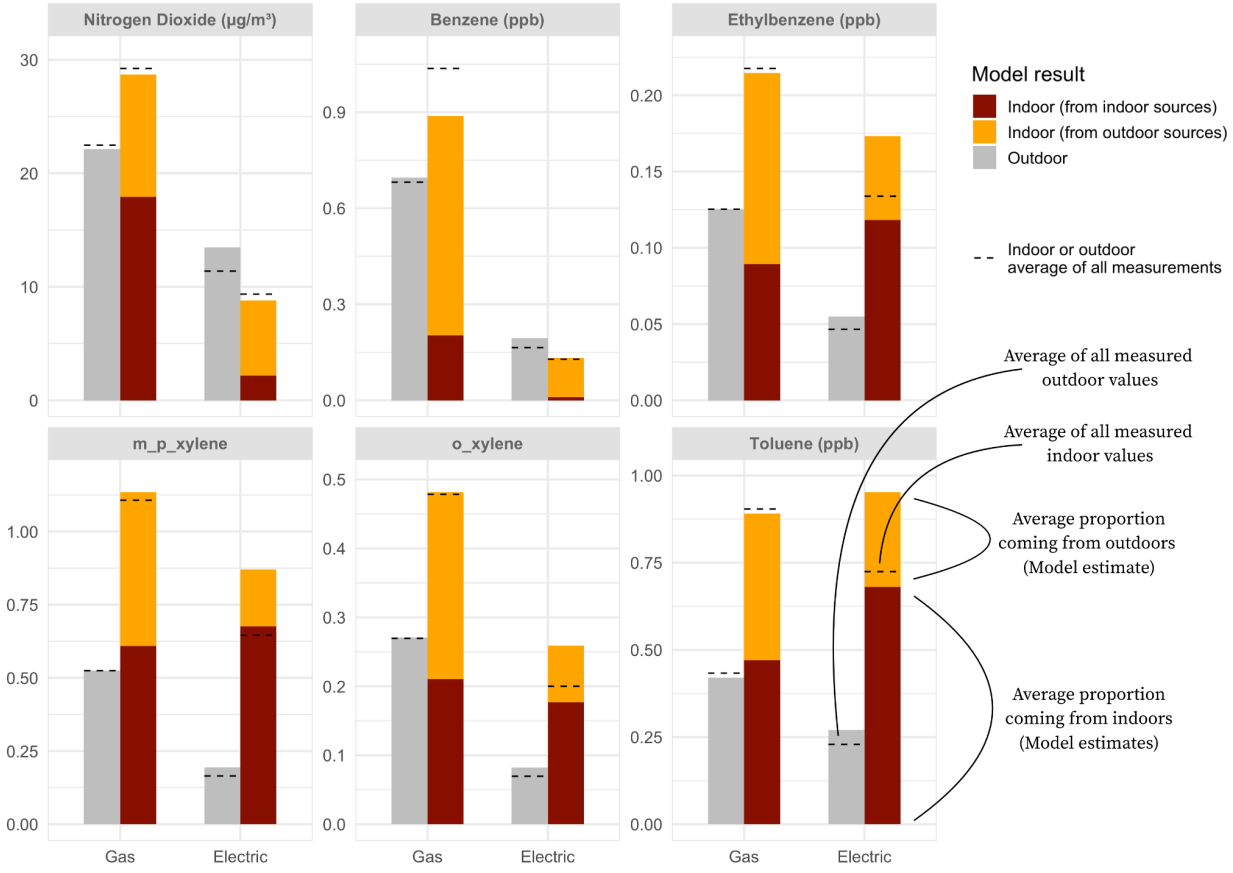

Figure 3. Average proportions of modelled outdoor and indoor concentrations of NO_2 (µg/m^3), benzene, toluene, ethylbenzene, m/p-xylene and o-xylene (ppb) for Phase 1 (Gas) and Phase 2 (Electric) across all households. Dashed lines show average

measured concentrations. The average of modelled outdoor and indoor-modelled values do not sum to the average of the measured values.

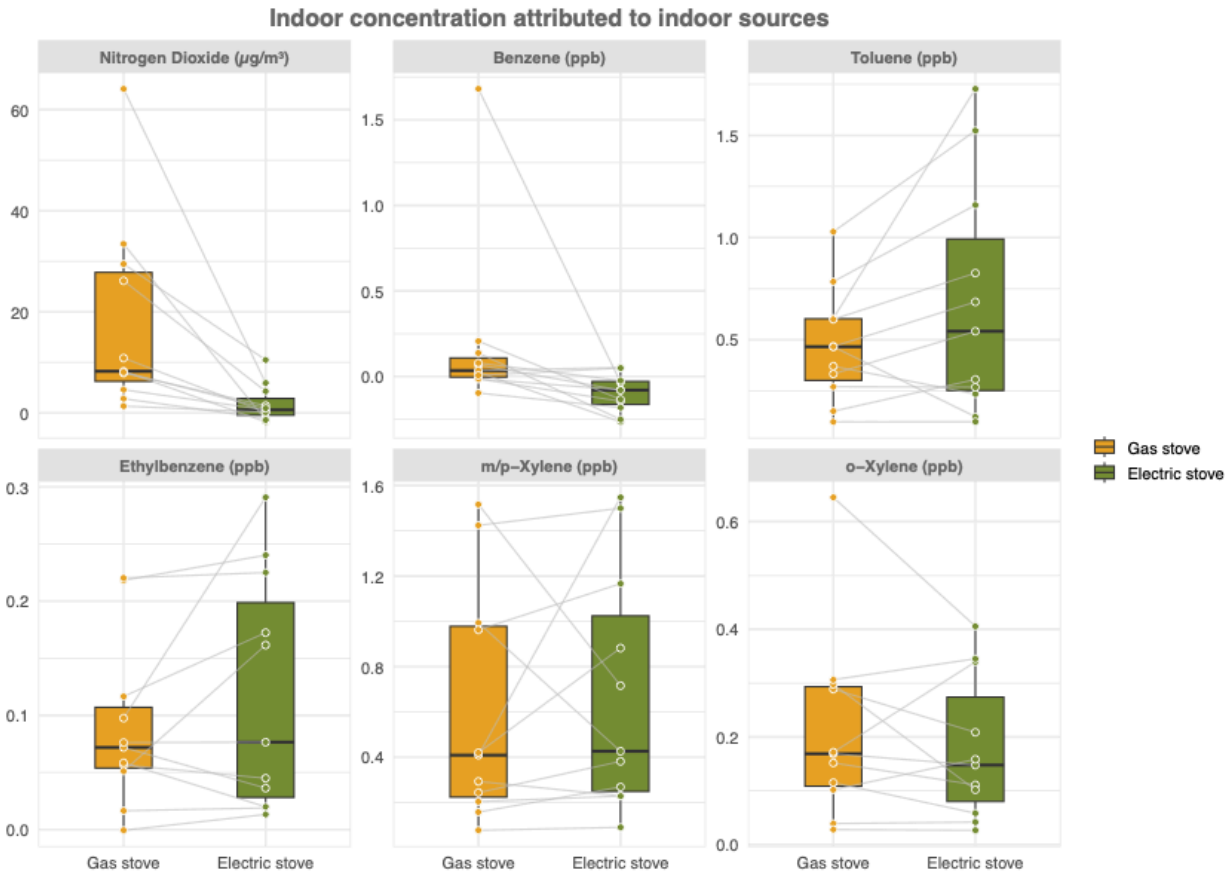

Figure 4. Indoor-model estimated concentrations of NO_2 (µg/m^3), benzene, toluene, ethylbenzene, m/p-xylene and o-xylene (ppb) for Phase 1 (Gas stove) and Phase 2 (Electric stove). Lines connect measurements made in the same homes.

Discussion

Greater indoor than outdoor reductions in NO_2 and benzene concentrations across the majority of households between Phase 1 and Phase 2 strongly suggests that removal of the gas cooking stove contributed to improved air quality. Model based estimates of the contribution of indoor sources to indoor NO_2 and benzene concentrations further support this conclusion (Figure 4).

For the remaining VOC species investigated there is no clear evidence that the removal of the gas cooking stove contributed to reduced concentrations. It is likely that concentrations of toluene, ethylbenzene and xylene are additionally controlled by emissions from a variety of non-gas sources, such as the use of solvents or cleaning products for example.

Because gas cookers are a source of BTEX emissions (incomplete combustion and leakage of gas), removing a gas cooker eliminates that source, such that indoor concentrations might be expected to reduce. The size of the indoor concentration change is, however, also dependent on other indoor sources, as well as indoor 'sinks' (e.g. deposition on to or absorption by household surfaces) and exchange with outdoor air. For those chemical species with minimal non-cooking sources, concentrations were expected to fall, and this can be observed in measured changes in benzene concentration between phases 1 and 2. Benzene sources indoors include fossil gas use, tobacco smoke (eliminated from the study sample) and some stored fuels and solvents. Toluene, ethylbenzene and xylenes are additionally found in paints and cleaning agents, air fresheners and building materials. Cleaning agents can be a significant source in kitchens in particular making a reduction in other BTEX species from gas stove removal alone less likely. We conclude that post-intervention concentrations of these species are more dependent on non-cooking sources.

It is also possible that the gas to electric stove replacement modified indoor air chemistry and pollutant sinks relevant to BTEX. The principal route for removal of BTEX species is oxidation by OH radicals, and to some extent also nitrate radicals and ozone. Indoor sources of OH radicals include reaction of ozone and VOCs, nitrous acid (HONO) photolysis and NOx chemistry. Since the replacement of the cooker removed a source of NOx and HONO, it is reasonable to assume that indoor OH radical production was reduced. This could slow the removal of BTEX species by oxidation. However it is likely that ventilation and to a lesser extent deposition are the dominant removal mechanisms, because BTEX chemical lifetimes are longer than air exchange times (Pagonis et al., 2019). We are not aware of any experimental evidence linking gas stove removal to increases in indoor BTEX concentrations.

Actions to reduce public health risks could include the provision of financial support to replace gas cooking equipment with electric alternatives, prohibiting the installation of gas stoves in new residential buildings, and indoor air quality regulations for NO_2 and health relevant VOCs. Priorities for financial support should be homes where vulnerable groups including children or those with existing health conditions are present, poorly ventilated or overcrowded homes and homes on low incomes.

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Appendix 1: Model results visualisation