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Did policies to abate atmospheric emissions from traffic have a positive effect in London?☆

Anna Font*, Gary W. Fuller

Environmental Research Group, MRC PHE Centre for Environment and Health, King's College London, London, SE1 9NH, United Kingdom

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ABSTRACT

A large number of policy initiatives are being taken at the European level, across the United Kingdom and in London to improve air quality and reduce population exposure to harmful pollutants from traffic emissions. Trends in roadside increments of nitrogen oxides (NO_X), nitrogen dioxide (NO_2), particulate matter (PM), black carbon (CBLK) and carbon dioxide (CO_2) were examined at 65 London monitoring sites for two periods of time: 2005–2009 and 2010–2014. Between 2005 and 2009 there was an overall increase in NO_2 reflecting the growing evidence of real world emissions from diesel vehicles. Conversely, NO_2 decreased by 10%·year^{-1} from 2010 onwards along with $PM_{2.5}$ (−28%·year^{-1}) and black carbon (−11%·year^{-1}). Downwards trends in air pollutants were not fully explained by changes in traffic counts therefore traffic exhaust emission abatement policies were proved to be successful in some locations. PM_{10} concentrations showed no significant overall change suggesting an increase in coarse particles which offset the decrease in tailpipe emissions; this was especially the case on roads in outer London where an increase in the number of Heavy Good Vehicles (HGVs) was seen. The majority of roads with increasing NO_X experienced an increase in buses and coaches. Changes in CO_2 from 2010 onwards did not match the downward predictions from reduced traffic flows and improved fleet efficiency. CO_2 increased along with increasing HGVs and buses. Polices to manage air pollution provided differential benefits across London's road network. To investigate this, *k*-means clustering technique was applied to group roads which behaved similarly in terms of trends to evaluate the effectiveness of policies to mitigate traffic emissions. This is the first time that London's roadside monitoring sites have been considered as a population rather than summarized as a mean behaviour only, allowing greater insight into the differential changes in air pollution abatement policies.

1. Introduction

The air pollution close to roads in large urban areas is usually affected by emissions from traffic such as nitrogen oxides (NO_X), particulate matter (PM) and black carbon (CBLK) among other pollutants (Sundvor et al., 2012). Tail-pipe emissions from traffic are dominated by diesel engines which emit NO_X in form of nitrogen monoxide (NO) and primary nitrogen dioxide (NO_2) (Carslaw et al., 2011) causing steep spatial gradients close to roads (Carslaw and Beevers, 2005). Diesel vehicles also emit elemental carbon (EC) and CBLK (Mansfield et al., 1991), fine and ultrafine particles (particles with <2.5 μm and <0.1 μm in diameter, respectively) which can be inhaled deeper into the lung and therefore are thought to be more toxic than larger particles (HEI, 2013). Moreover, other non-exhaust traffic-related emissions such as resuspension, tyre-wear and brake-wear could represent an important fraction of coarse PM on roads (particles with >2.5 μm) (Amato et al., 2016). Exposure to traffic-related pollutants can be very considerable alongside urban roads in central areas, along retail and popular streets used by pedestrians. Adverse health effects associated with proximity to roads have been observed due to higher concentrations of individual or combinations of traffic-related pollutants (WHO, 2013). The mortality burden of NO_2 and $PM_{2.5}$ in London in 2010 has been estimated as equivalent to 9416 deaths at typical ages (Walton et al., 2015). Recent studies have found positive associations between EC and CBLK from diesel exhaust emissions and respiratory mortality (Atkinson et al., 2015). Toxicological research increasingly indicates that non-exhaust pollutants could also be responsible for some of the observed

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* Corresponding author.
E-mail address: anna.font_font@kcl.ac.uk (A. Font).

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health effects (WHO, 2013). Traffic sources also release gases including carbon dioxide (CO_2) that have an impact on climate (IPCC, 2013). Anthropogenic sources of CO_2 in urban areas are related to combustion processes such as burning of fossil fuels and electricity production. Roadside increments of CO_2 can be taken as indication of direct exhaust emissions from traffic (Gratani and Varone, 2014).

A suite of policies have been implemented to improve air quality and reduce population exposure. The Euro emission standards were introduced in the early 1990s to reduce exhaust emissions from new vehicles and tighter standards have been introduced in the last two decades. Transport for London (TfL) invested in a program to fit a catalytic diesel particulate filters (CDPF) to its older buses by the end of 2005 (Carslaw and Beevers, 2005). By 2014 TfL completed a second bus retrofit campaign with over 1000 Euro III buses fitted with a Selective Catalytic Reduction (SCR) system to reduce NO_X emissions. Fitting SCR was prioritized for those buses with routes along busy roads in central London (TfL, 2014a). Other initiatives across London include the Low Emission Zone implemented in 2008 which limited the entrance of the most polluting diesel Heavy Good Vehicles (HGVs) in London. The Mayor's Air Quality Strategy in 2010 planned the roll out of new hybrid buses and low-emission buses (Euro IV) (GLA, 2010). All these policies have been accompanied by many local-scale schemes implemented by the London's boroughs. The ensemble of these initiatives is expected to have a direct impact on the air quality in the whole of London but especially alongside busy roads. Whilst the vast majority of roadside locations in London met the PM_{10} EU Annual Mean Limit Value of 40 $\mu g\ m^{-3}$ in 2014, the majority still exceeded the NO_2 EU Limit Value of 40 $\mu g\ m^{-3}$ by a large margin (Mittal et al., 2016). Additionally, meeting the EU $PM_{2.5}$ exposure reduction target remains challenging.

Evaluating the success of policies in improving air quality can be done though the study of trends in atmospheric concentrations in time (e.g. Gualtieri et al., 2014). Most of the approaches calculate trends in concentrations using one monitoring site representative of a given exposure scenario (e.g. roadside site); or by averaging the concentrations from a group of similar sites. In large urban settings, such as London, trends in air quality monitoring sites have been classified by their distance to the city centre, for instance, inner and outer London (Beevers et al., 2012). Other studies considered the variability of trends observed across a network. Colette et al. (2015) calculated trends in air pollutants in Europe using the AIRBASE network and a probability density function to summarize the trend distribution. Other recent approaches include the estimation of trends in air pollutants from satellite-based observations. These benefit from their wide spatial coverage, allowing calculation of trends at the continental scale (van der A. et al., 2008), country-scale (e.g. Cuevas et al., 2014) or at various urban centres worldwide (Schneider et al., 2015). Although these methods are valid and useful, these approaches can mask a wide heterogeneity in the impact of policies across an urban area and especially in roadside locations where emissions have a large spatial and temporal variability.

Our study was designed to evaluate the success of the policies to reduce the air pollution concentrations in London with special focus on traffic emissions. We benefitted from the large number of monitoring stations in the urban agglomeration (more than 100 monitoring stations in 2014) and trends of air pollutants concentrations were calculated for all 65 roadside sites available between 2005 and 2014. Due to the spatial representativeness of air quality monitoring stations, the duration and diversity of data, the London air quality database offers an unprecedented and effective way to analyze trends in surface air pollutants concentrations. The overall trend of air pollutants in London was calculated using statistical approaches used in meta-analysis studies that consider individual and population-wide variability. Roadside locations were grouped according to recent changes in air pollution concentrations and trends were related to specific policies and to changes in traffic counts and composition. This approach would be applicable to other cities with a large network of monitoring sites, and also at the country, region and worldwide scale.

2. Materials and methods

2.1. Monitoring sites

Measurements of NO_X, NO_2, particulate matter with aerodynamic diameter <10 μm (PM_{10}) and <2.5 μm ($PM_{2.5}$), black carbon (CBLK) and carbon dioxide (CO_2) were extracted from the UK Automatic Urban and Rural Network (AURN) and the London Air Quality Network (LAQN). These comprised 65 roadside Air Quality Monitoring Sites (AQMSs) (Supplementary Fig. 1). Note that some sites had collocated PM instruments measuring by different methods, e.g. Marylebone Road where PM was monitored by both TEOM and TEOM-FDMS. To distinguish these apart, the methods were assigned to different site codes (i.e. MY1 and MY7, respectively). Three roadside AQMSs in the London network measured CO_2 and CBLK for the period 2010–2014.

Measurements from Kensington and Chelsea - North Kensington (KC, 51.521°N, −0.2135°E) were taken as background concentrations (Supplementary Fig. 1). KC was chosen as background for three reasons: i) the use of a single background site allowed roadside increments to be directly compared between different roadside locations; ii) it is the urban background AQMS with the longest complete time series for all pollutants; iii) trends observed at KC were the same (within 2σ confidence interval) of the overall trends observed for all urban background sites in London (Supplementary Figs. 2 and 3), with the only exception being trends for NO_2 between 2005 and 2009 when a faster decrease ($-1.07\ \mu g\ m^{-3}\ year^{-1}$) was observed compared with the other urban background sites ($-0.37\ \mu g\ m^{-3}\ year^{-1}$).

The distance to London's city centre was calculated for each AQMS, setting the centre at Charing Cross (51.508°N, 0.125°W). Sites <10 km from Charing Cross were considered inner London; sites >10 km away from Charing Cross were considered outer London.

2.2. Measurements

NO_X ($NO + NO_2$) was measured by chemi-luminiscence and fortnightly calibrations enabled the traceability of measurements to national metrological standards. PM_{10} and $PM_{2.5}$ were measured by TEOM-FDMS (Tapered Element Oscillating Microbalance - Filter Dynamics Measurement System); by TEOM and by MetOne BAM (Beta Attenuation Monitors). TEOM-FDMS measurements were considered equivalent to the EU reference method, which is based on 24-h sampling and gravimetric analysis. PM_{10} measurements made by TEOM were converted to reference equivalent using the Volatile Correction Model (VCM) (Green et al., 2009). $PM_{2.5}$ measurements by TEOM were not corrected to reference equivalent as there is currently no agreed method for this. PM measurements by BAM were corrected to EU Reference equivalent using a factor of 1/1.2 (DEFRA, 2010). CBLK in $PM_{2.5}$ was measured by the Magee Aethalometer AE22 and raw data was corrected for the filter loading effect (Virkkula et al., 2007; Butterfiled et al., 2013). All instruments were subject to twice yearly audit tests by the National Physical Laboratory or Ricardo AEA.

CO_2 concentrations were measured using a LiCOR-820 Non-Dispersive IR analyzer. Two-point calibrations were carried out

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