Air pollution: outdoor air quality and health

DRAFT Evidence review 1 on: Environmental change and development planning

Internal Guideline Development Team:
Hugo Crombie, Diana O'Rourke, Stephen Robinson

October 2016

National Institute for Health and Care Excellence

1 Introduction **6**

2 Methods **7**

2.1 Review questions 7

2.2 Searching, screening, data extraction and quality assessment 7

3 Results **10**

3.1 Flow of literature through the review 10

3.2 Characteristics of the included studies 11

3.2.1 Review question 2: Are interventions to develop public transport routes and services, effective and cost effective at reducing the health impact of, or people’s exposure to, traffic-related air pollution? 12

3.2.2 Review question 3: Are interventions to develop routes and infrastructure to support low emission modes of transport effective and cost effective at reducing the health impact of, or people’s exposure to, traffic-related air pollution? 12

3.2.3 Review question 4: Are measures to promote absorption, adsorption or impingement deposition, and catalytic action effective and cost effective at reducing the health impact of, or people’s exposure to, traffic-related air pollution? 14

4 Study Findings **15**

4.1 Review question 2: Are interventions to develop public transport routes and services, effective and cost effective at reducing the health impact of, or people’s exposure to, traffic-related air pollution? 15

4.1.1 Changing fuel use and operational practices of buses 15

4.1.2 Construction of a contra flow bus lane 17

4.2 Review question 3: Are interventions to develop routes and infrastructure to support low emission modes of transport effective and cost effective at reducing the health impact of, or people’s exposure to, traffic-related air pollution? 18

4.2.1 Cycle routes 18

4.2.2 Public transport 27

4.2.3 Construction of a bypass 40

4.3 Review question 4: Are measures to promote absorption, adsorption or impingement deposition, and catalytic action effective and cost effective at reducing the health impact of, or people’s exposure to, traffic-related air pollution? 41

4.3.1 Use of natural and artificial barriers 42

4.3.2 Use of dust suppressants 48

4.3.3 Use of street washing and sweeping 50

4.3.4 Urban greening 50

5 Discussion **55**

5.1 Strengths and limitations of the review 55

5.2 Applicability 56

5.3 Gaps in the evidence 57

5.3.1 Review question 2: Are interventions to develop public transport routes and services, effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution? 57

5.3.2 Review question 3: Are interventions to develop routes and infrastructure to support low emission modes of transport effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution? 57

5.3.3 Review question 4: Are measures to promote absorption, adsorption or impingement deposition, and catalytic action effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution? 58

6 Included studies 59

Appendix 1 Evidence Tables

Appendix 2 Quality of included studies

Appendix 3 Search strategy

Appendix 4 Excluded studies

Appendix 5 Quality Appraisal checklists

Appendix 6 Review Protocols

1 Introduction

This evidence review is the first of a series conducted to support the development of guideline on road transport related air pollution. The scope for the guideline is available here.

This review addresses Topic 1: Environmental change and development planning, as set out in the review protocols. The following questions were included in this topic:

- Review question 1: Are planning development control decisions and interventions effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?
- Review question 2: Are interventions to develop public transport routes and services, effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?
- Review question 3: Are interventions to develop routes and infrastructure to support low emission modes of transport effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?
- Review question 4: Are measures to promote absorption, adsorption or impingement deposition, and catalytic action effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

Questions 5-12 are addressed in reviews 2 and 3. Where cost effectiveness studies were identified they are included with the relevant question.

No included studies were identified for review question 1, this review covers only review questions 2-4.

2 Methods

This review was conducted according to the methods set out in Developing NICE guidelines: the manual (NICE 2014).

2.1 Review questions

Review question 1: Are planning development control decisions and interventions effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

Review question 2: Are interventions to develop public transport routes and services, effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

Review question 3: Are interventions to develop routes and infrastructure to support low emission modes of transport effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

Review question 4: Are measures to promote absorption, adsorption or impingement deposition, and catalytic action effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

2.2 Searching, screening, data extraction and quality assessment

A systematic search of relevant databases and websites was conducted from September 1995 (this start date was chosen as it corresponds with the passing of the Environment Act which provided the legislative base for the National Air Quality Strategy) to September 2015. Updated searches from September 2015 will be considered as indicated in the manual before finalising the guideline (see Appendix 3).

A call for evidence from stakeholders was also undertaken to help identify additional relevant evidence. This resulted in 34 unique studies being added to the search results).

The protocols outlining the methods for the review, including the search protocols and methods for data screening, quality assessment and synthesis are available (see Appendix 6).

All references from the database searches and call for evidence were screened on title and abstract against the criteria set out in the protocols. A random sample of 10% of titles and abstracts was screened by 2 reviewers independently, with differences resolved by discussion. Agreement at this stage was 97.3%. Full-text screening was carried out by two reviewers independently on 10% of papers. Agreement at this stage was 93%. Any differences were resolved by discussion. Reasons for exclusion at full paper stage were recorded (see Appendix 4).

Systematic reviews were used to identify potentially relevant primary studies. The references from these were assessed against the criteria set out in the protocols.

Modelling studies were identified at all stages and assessed against the criteria in the protocols. Modelling studies for questions where there was insufficient evidence from comparative studies were included in the review. Decisions about whether there was sufficient evidence were made in discussion with the PHAC. All cost effectiveness studies were also included.

Each included study was data extracted by one reviewer, with all data checked in detail by a second reviewer. Any differences were resolved by discussion.

Included studies were rated individually to indicate their quality, based on assessment using a checklist. Each included study was assessed by one reviewer and checked by another. Any differences in quality rating were resolved by discussion. The tools used to assess the quality of studies are included in Appendix 5 and a summary of the critical appraisal results of all included studies is included in Appendix 2. The quality ratings used were:

++ All or most of the checklist criteria have been fulfilled, and where they have not been fulfilled the conclusions are very unlikely to alter.

+ Some of the checklist criteria have been fulfilled, and where they have not been fulfilled, or are not adequately described, the conclusions are unlikely to alter.

– Few or no checklist criteria have been fulfilled and the conclusions are likely or very likely to alter.

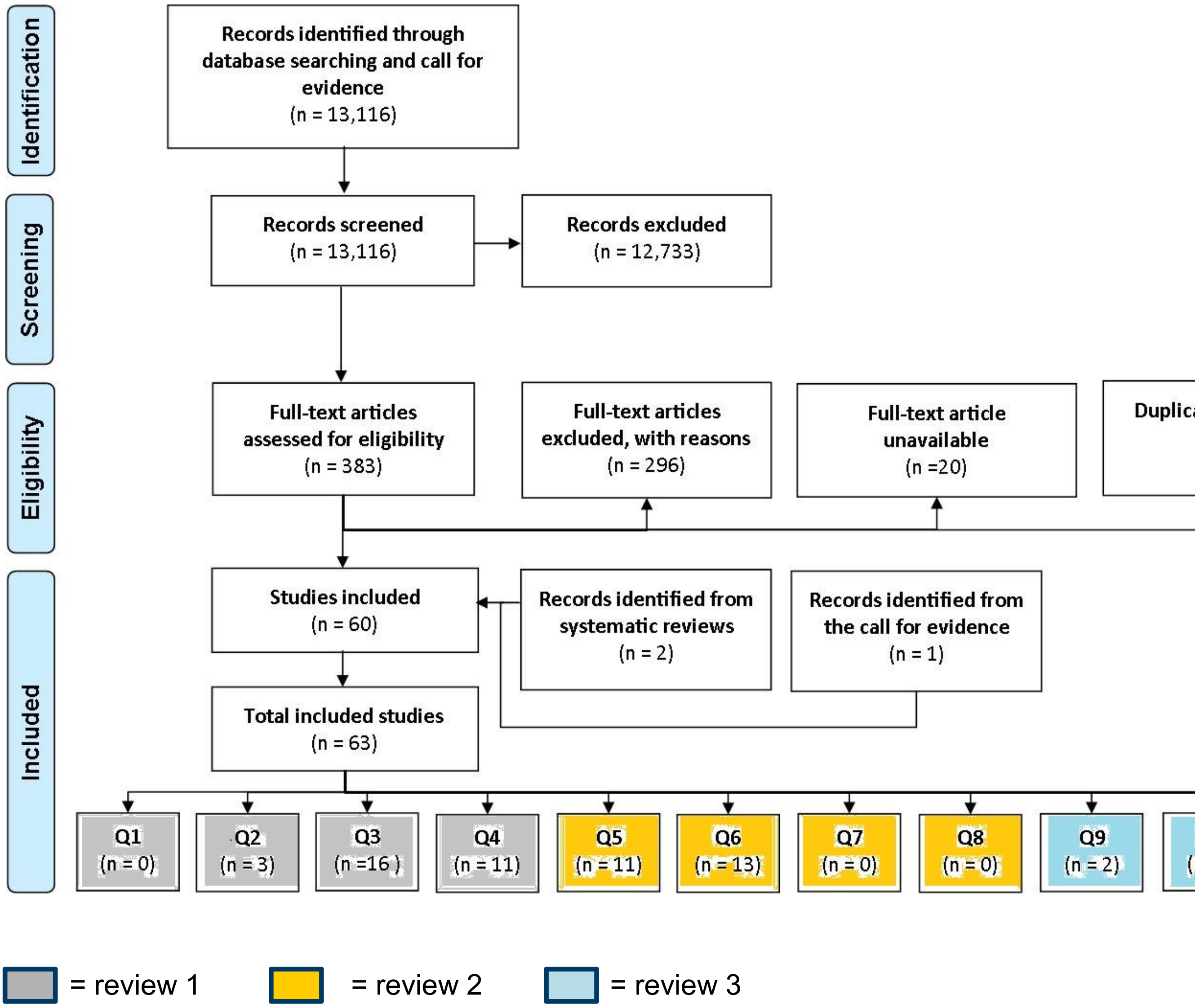
3 Results

3.1 Flow of literature through the review

Thirty studies were included in the current evidence review. The flow of literature through the reviews is summarised in Figure 1. A brief summary of reasons for exclusion at full text for all reviews is included in the table below.

Reason	**Number**
Did not meet the study type criteria	30
Conference abstract	18
Modelling study	78
No relevant intervention	57
Out of scope	41
Outcomes not relevant	54
Country not in the protocol	1
Published outside search dates	1
Other	14
Total	**294**

Figure 1: Flow of literature through the review



Three cost effectiveness studies are included in review 1 (question 3); 2 in review 2 (question 6). No cost effectiveness studies are included in review 3.

3.2 Characteristics of the included studies

Full details of the included studies are given in the evidence tables in Appendix 1. Table 3.2.1 (RQ2), Table 3.2.2 (RQ3) and Table 3.2.3 (RQ4) below shows in which country the studies were conducted, and provides a brief summary of the interventions and settings investigated in these studies.

3.2.1 Review question 2: Are interventions to develop public transport routes and services, effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

First author, year	Design	Country	Setting	Intervention	QA rating
Alam, 2014a	Modelling study	Canada	General urban	Changing fuel use of buses	-
Alam, 2014b	Modelling study	Canada	General urban	Bus service changes	-
Stamos, 2013	Modelling study	Greece	General urban	Construction of a contra flow bus lane	-

3.2.2 Review question 3: Are interventions to develop routes and infrastructure to support low emission modes of transport effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

Effectiveness / Modelling

First author, year	Design	Country	Setting	Intervention	QA rating
Cycle route infrastructure					
Bean, 2011	Comparative study	UK	On and off-road cycle routes	Cycling on vs off road route	-
Boogaard, 2009	Non-randomised controlled trial	Netherlands	Onroad	Cycling routes	-
Hatzopoulou, 2013	Comparative study	Canada	On-road	Cycling routes and distance from road	-
Jarjour, 2013	Non-controlled comparative study	USA	On-road	Cycling routes	-
Kendrick, 2009	Comparative study	USA	On road	Cycling routes	-
MacNaughton, 2014	Comparative study	USA	On-road	Cycling routes	-
Public transport					
Burgard, 2009	Non-randomised controlled trial	USA	On road	School buses with oxidative catalysts and particulate traps	-

Gramsch, 2013	Controlled before and after	Chile	On-road	Public transportation system	-
Bypass Construction					
Burr, 2004	Controlled before and after	UK	Residential / commercial streets	Construction of a bypass	-
Alternative fuel interventions					
Chong, 2014	Modelling study	UK	General urban	alternative fuel vehicles	-
Goncalves, 2009a	Modelling study	Spain	General urban	alternative fuel vehicles	-
Goncalves, 2009b	Modelling study	Spain	General urban	alternative fuel vehicles	-
Soret, 2014	Modelling study	Spain	General urban	alternative fuel vehicles	-

Cost Effectiveness (alternative fuel interventions – only)

Author, year	Country	Setting	Population	Intervention/comparator	Outcomes	QA rating
Cohen et al 2003	US	Hypothetical urban area	Exposed population (general)	Urban public transport fuel choice: 1. CD – conventional diesel 2. ECD – emission controlled diesel 3. CNG – compressed natural gas	ICER/QALYs	+
Cohen 2005	US	Three hypothetical school districts differing in size, population density and distance the bus travels per annum	Exposed population (general)	School bus fuel choice: 1. CD – conventional diesel 2. ECD – emission controlled diesel 3. CNG – compressed natural gas	ICER/QALYs	+
Krutilla and Graham 2012	US	Adoption of diesel-electric hybrids in urban delivery vehicles		Adoption of diesel-electric hybrids in urban delivery vehicles	Net Present Value	+

3.2.3 Review question 4: Are measures to promote absorption, adsorption or impingement deposition, and catalytic action effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

First author, year	Design	Country	Setting	Intervention	QA rating
Use of natural and artificial barriers					
Al-Dabbous et al., 2014	Controlled study	UK	Roadside adjacent to a 4 lane highway	Vegetation barrier	-
Brantley et al., 2013	Controlled study	USA	Roadside area adjacent to a six-lane highway	Vegetation barrier	-
Hagler et al., 2012	Controlled study	USA	Roadside and near road locations	Roadside and vegetation barriers	-
Ning et al., 2010	Controlled trial	USA	Two major freeways	Roadside noise barrier	-
Baldauf et al 2016	Controlled trial	USA	Adjacent to large highway	Roadside noise barrier	-
Use of dust suppressants					
Amato et al., 2014	Controlled before and after study	Spain	2.5km of trafficked road in a commercial district	Dust suppressants	-
Gillies et al., 1999	Controlled study	USA	Unpaved stretch of road	Dust suppressants	-
Street washing and sweeping					
Amato et al., 2009	Controlled study	Spain	A commercial and residential street	Street washing	-
Urban greening					
Pugh, 2012	Modelling study	UK	General urban	Urban greening	+
Vos, 2013	Modelling study	Belgium	General urban	Urban greening	+
Vranckx, 2015	Modelling study	Belgium	General urban	Urban greening	+

4 Study Findings

4.1 Review question 2: Are interventions to develop public transport routes and services, effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

Three modelling studies were included in the review. Overall, the quality of the studies was poor with all studies graded [-] (see Table 3.2.1).

Studies were grouped by the intervention the study assessed:

- Changing fuel use and operational practices of buses
- Construction of a contra flow bus lane

4.1.1 Changing fuel use and operational practices of buses

A [-] quality modelling study by **Alam (2014a)**[1] in Montreal looked at the impact of changing the fuel use of buses and five combinations of changes to operational practices (traffic signal priority, bus stop relocation and 'queue jumper lanes'), all with and without fuel use changes (diesel and compressed natural gas (CNG)).

The study corridor in Montreal is about 5.1km (3.2 miles) with various gradients ranging from -17% to +8%. It has a high frequency of buses (4-5 min) during peak periods. There are significant differences in traffic flow between the north bound and southbound directions and between morning and afternoon peak periods. The MOVES model[1] was used to estimate the emissions of $PM_{2.5}$ for each segment of the route (using all the buses that run along the corridor in the morning peak period). Emissions were estimated separately for the north-bound and south-bound directions and for both running and idling.

Following the base case traffic and emission simulations, replacement of the current Ultra low sulphur diesel (ULSD) with compressed natural gas (CNG) was simulated.

Five operational scenarios were simulated using ULSD and CNG in order to identify the additional impact of an alternative technology under various bus operations. The scenarios were:

Scenario 1 – Transit signal priority (TSP) to extend the green phase to reduce idling at the bus stop.

[1] Motor Vehicle Emissions Simulator

Scenario 2 – relocation of bus-stops to mid-block without TSP to avoid passengers waiting in areas of higher air pollutant concentrations at intersections.

Scenario 3 – relocation of bus-stop to mid-block plus TSP.

Scenario 4 – queue jumper lane without TSP. Queue jumper lanes (to provide buses with a head start and avoid queueing behind right turning traffic) were introduced at each intersection without relocating bus-stops.

Scenario 5 – queue jumper lane, relocation of bus-stop and TSP strategy. *This scenario combines all the previous improvements under one scenario.*

Switching fuel from diesel to CNG under current bus operations reduces emissions of $PM_{2.5}$ by 89% in the less congested northbound (NB) direction and by 85% in the congested southbound (SB) direction.

Table 1. Changes in $PM_{2.5}$ emissions for different fuels and scenarios.

Scenario	$PM_{2.5}$ (g/mile bus) diesel (% change)		$PM_{2.5}$ (g/mile bus) CNG (% change)	
	Southbound	Northbound	Southbound	Northbound
Base case	0.046	0.036	0.007 (-84.79%)	0.004 (-88.68%)
1 – TSP	-16%	-8.76%	-92.4%	-90.12%
2 – bus stop relocation	-9.89%	0.00%	-92.4%	-88.68%
3 – bus stop relocation + TSP	-16.25%	-0.73%	-92.4%	-88.68%
4 – queue jumper lane	-17.56%	-1.40%	-92.4%	-88.77%
5 – TSP + bus stop relocation + queue jumper lane	-18.36%	-2.06%	-92.69%	-88.77%

Operational changes alone produced reductions in $PM_{2.5}$ of between 0 and 18%.

Operational changes plus fuel switch produced reductions in $PM_{2.5}$ of between 89% and 93%.

A second [-] quality modelling study by **Alam (2014b)[2]** also looks at buses using a route in Montreal. The aim was to examine the effect of isolated and combined transit service improvements on pollutants (including $PM_{2.5}$ and NO_x).

The study examined a busy transit corridor in Montreal, running 5.8 miles (9.3km) including 28 signalised intersections equipped with Transit Signal Priority (TSP) system which provides an extended green light phase or a shorter red light on the approach of a bus.

The study quantified the effects of bus service changes on bus emissions. Service changes were:

- Use of smart cards.

- A limited-stop bus express service running parallel to the original route serving 40% of the bus stops on weekdays (from 06:00 to 19:00).
- Use of reserved bus lanes during peak periods.

Emissions generated during bus operations are estimated using the MOVES model, developed by the US Environmental Protection Agency using second-by-second speeds. Emissions were estimated for fine particulate matter ($PM_{2.5}$) at the segment level (including running and idling) and stop-level (only idling).

The largest positive impact on emissions is associated with the introduction of Reserved Bus Lanes which can reduce $PM_{2.5}$ emissions by 4.4mg/mile of bus travel. The express bus service has the second largest negative coefficient, decreasing $PM_{2.5}$ by 4.3mg/bus mile.

4.1.2 Construction of a contra flow bus lane

A [-] quality modelling study by **Stamos (2013)[27]** in Thessaloniki (Greece) examined the potential impact of a planned contra flow bus lane. This would be situated on a 0.9km stretch of a four lane, one-way road (one lane was used as a designated contra-flow bus lane, moving it from its current site in an adjacent street) in the central business district.

The model used (Simulation and Assignment of Traffic to Urban Road Networks - SATURN[2]) is a traffic simulation model covering the whole of the central area.

The modelling found that traffic speeds decreased (from 7.87 to 7.11kph) and congestion increased (from average delays of 599s to 675s). Nitrogen dioxide emissions are not changed. Changes to traffic in the zone around the street in which the contraflow lane was planned (the buffer zone) vary. NO_2 emissions in the buffer zone decrease by around 2.1% (from 95 to 93 kg). The authors note that a shift to public transportation might reduce emissions, although this is not included in the current model.

Modelling Evidence statement 2.1 Bus operations

Weak evidence from 2 [-] quality modelling studies from Canada[1,2] and 1 [-] modelling study from Greece looked at the impact of changes to bus operations on air pollutants. Two studies from Canada[1,2] showed that the modelled impact on $PM_{2.5}$ from operational changes alone are modest (0-18%)[1] (a reduction of 0.03781g/bus mile) compared to reductions from conversion to compressed natural gas (modelled reductions of 85-93% (reductions of between 0.007042 and 0.003488g/bus mile))[2].

[2] https://trid.trb.org/view.aspx?id=491629

Introduction of reserved bus lanes and an express bus service reduced $PM_{2.5}$ emissions by 4.4 and 4.3mg/bus mile, respectively[1].

One modelling study from Greece showed that an intervention to reallocate road space to a contraflow bus lane could increase congestion and decrease speed. This will influence emissions of nitrogen oxides and hydrocarbons and fuel consumption[3]. The direction and extent of the impact will depend on local traffic factors.

1. Alam 2014a (−)

2. Alam 2014b (−)

27 Stamos 2013 (−)

Applicability. The studies come from Canada and Greece. Potential differences in the composition of the vehicle fleet and local traffic systems compared with the UK means that they are partially applicable.

4.2 Review question 3: Are interventions to develop routes and infrastructure to support low emission modes of transport effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

Thirteen studies were included in the review. Overall, the quality of the studies was poor with all studies graded [-] (see Table 3.2.1).

Studies were grouped by the intervention the study assessed:

- Cycle infrastructure (6 studies)
- Alterations to public transport (6 studies)
- Use of alternative fuels (4 modelling studies, 2 cost effectiveness studies)
- Construction of a bypass (1 study)

4.2.1 Cycle infrastructure (on and off road routes, cycle lanes and paths)

Bean et al (2011, -)[6] examined three cycle journeys to the University of York in a comparative study. Each journey had two versions, one 'on road' and one 'off road'. Measurements of NO_2 concentrations along all six routes were made with diffusion tubes in August and September 2008.

The three routes were:

- Route 1 **on-road:** 7 km long taking about 20 minutes including busy roads and the city centre. **Off-road:** 7.8 km taking about 23.5 minutes with off-road (approx 58% of the time) or less trafficked roads.
- Route 2 **on-road:** 3.5 km long and took about 10 minutes, joining the middle stages of Route 1 after 2.5 minutes. **Off-road:** 4.5 km long and took about 15 minutes to cycle; 66% of this journey was on designated off-road cycle paths.
- Route 3 **on-road:** 8.6 km and took about 23 minutes to cycle. **Off-road:** 8.7 km long and took about 24 minutes; 76% on designated off-road cycle paths.

For each of the three routes, NO_2 concentrations were higher overall for the on-road compared to the off-road route.

As well as average concentration, exposure was estimated by multiplying by the time spent travelling. Cycling the off-road rather than the on-road route decreased the average time-weighted concentration of NO_2 by between 29 (14.3-10.2ppb) and 41% (17.1-10.1ppb), with a mean of 37.5% (35%, 14.7-9.5ppb, August; 40%, 16.2-9.8ppb, September). The difference between average concentrations on the on-road route and the equivalent off-road route was significant (t = 11.78; P < 0.01).

A paired t-test showed that the mean difference in exposure between on-road and off-road routes was significant (t = 3.50; P = 0.017) actual levels not reported. Therefore, there was a significant reduction in the exposure a cyclist receives by taking an off-road route, even though taking the off-road route for any journey in this study extended the exposure time.

The authors note that the difference in effect between time-weighted concentration and exposure will be dependent on differences in journey times for both routes.

Hatzopoulou (2013[19], -) evaluated personal exposures to multiple air pollutants among cyclists in Montreal, Canada, in a comparative study. It also looked at determinants of exposure (such as type of cycling lane (separated vs non-separated), distance of the cycling lane from the nearest traffic lane, traffic counts and characteristics (i.e., petrol vs diesel vehicles), time-period of cycling (morning/evening commute), and meteorological variables).

Cycling lanes were categorized as either separated (including fully separated lanes and lanes separated by parked cars) or not separated (on-road lanes with no physical barrier between the cycling lane and traffic).

Ultra-fine particles (UFP) (0.01–1 mm), $PM_{2.5}$ and black carbon (BC) were measured real-time using monitoring equipment in panniers located on each bicycle and tubes attached to the air intakes of each device were run along bicycle frames to sample as close as possible to participants' breathing zones.

Routes ranged from 16 to 19 kilometers in length and the same route was traversed each day in both the morning and evening rush-hour time-periods.

Traffic counts and characteristics (number of diesel and non-diesel vehicles/min) were recorded at 79 intersections along the cycling routes. Traffic counts were based on 10-min observations during each time-period and distances from the centre of bicycle lanes (either on-road or separated) to the centre of adjacent roadways were determined at each traffic count point.

On average, personal air pollution exposures were similar during the morning and evening commutes with UFP and BC exposures tending to be higher during the morning commute

Use of separated cycling lanes was associated with small decreases in exposure to UFPs (-9.2%) and BC (-9%) and increased exposure to $PM_{2.5}$ (2%). For BC, the author suggested this magnitude of this decrease was equivalent to removing eight diesel vehicles per hour from the nearest traffic lane. Findings examining the impact of cycling lane distance from the nearest traffic lane were generally consistent with findings for cycling lane type. Each 5-m separation of the cycling lane was associated with modest decreases in exposure of 1.2% (95% CI: -10, 8.5) for UFPs, 2.5% (95% CI: -17, 12) for BC, and 3.6% (95% CI: -9.5, 1.9) for CO, adjusted for traffic counts and meteorological factors. The same separation was associated with a 3.5% (95% CI: -9.1, 18) increase in personal exposure to $PM_{2.5}$.

Clear explanations for the increased exposure to $PM_{2.5}$ were not reported, although the authors suggested proximity to sources may be less important for $PM_{2.5}$ than the other pollutants owing to the regional nature of this pollutant.

Jarjour et al (2013[20], -) examined exposure to air pollution among 15 adults cycling low or high traffic routes in Berkeley in a non-controlled comparative study. The routes were designed to be similar in length (8-9.5km) and in elevation change (61m). The high traffic route was on roads with between 10,000-26,000 vehicles per day (VPD). Traffic count data was not available for the roads on the low traffic route but the authors note that they were likely to be below 3-4,000 VPD, and below 1,500 VPD in many parts.

Participants were regular cyclists, aged 23-48 with no existing respiratory, cardiovascular or other chronic health conditions. UFP, black carbon and $PM_{2.5}$ were measured using monitors in the rear basket of the bike. All participants completed both routes.

Lung function was examined using forced vital capacity (FVC), forced expiratory volume in one second (FEV1), their ratio (FEV1/FVC) and forced expiratory flow between 25-75% of vital capacity. All changes in lung function were clinically insignificant and paired *t*-tests by subject found no significant p-values. The authors note that participants were selected to have no pre-existing chronic cardiovascular conditions (including asthma).

Table 2. Pollutant exposures for low and high traffic routes

Pollutant		Mean	Minimum	Maximum	5-95 %ile
UFP	Low traffic	14,311	2,771	376,495	4,621 to 29,882
	High traffic	18,545	1,900	1,033,188	4,148 to 51,265
$PM_{2.5}$	Low traffic	4.88	2.25	20.96	2.65 to 6.84
	High traffic	5.12	2.25	27.40	2.94 to 7.10
BC	Low traffic	1.76	0.11	63.83	0.50 to 4.03
	High traffic	2.06	0.1	53.53	0.37 to 5.06

UFP – particles/cm^3, BC and $PM_{2.5}$ - $\mu g/m^3$

Table 3. Paired t-test by subject, average exposure for each subject's high vs low traffic ride average

Pollutant	Mean		p-value	95% CI of difference
	Low traffic	High traffic		
UFP	13,517.13	19,945.34	0.01	1,656.84 to 11,199.58
$PM_{2.5}$	4.88	4.53	0.60	-1.90 to 1.19
BC	1.73	2.10	0.06	-0.02 to 0.26

UFP – particles/cm^3, BC and $PM_{2.5}$ - $\mu g/m^3$

Exposure to $PM_{2.5}$ was lower in the high-traffic route; this was not statistically significant. Exposure to UFPs and BC were higher in the high-traffic route, for UFP this reached a significant level, and for BC was 'borderline significant'. The authors note that the result for $PM_{2.5}$ would be expected for a pollutant with lower variability at small spatial scales. Higher levels of pollutants on the low-traffic route were generally seen where the route crossed main roads (levels were not reported).

MacNaughton et al (2014[23], -) looked at the impact of bicycle route type on cyclists' exposure to traffic related air pollution in a comparative study.

The bike routes were categorized into three types:

- Bike paths: separated from motor traffic;
- Bike lanes: adjacent to motor traffic;
- Designated bike lanes: lanes shared by buses and cyclists.

Exposure to NO_2 and BC using each bike route type in Boston, Massachusetts was measured during morning and evening commute (07:00 to 10:00) and (15:00 to 18:00) respectively. Five bike routes were selected to represent travel over a variety of bike route types during variable traffic and atmospheric conditions and each monitored 4 times.

Associations between route type and pollution levels were examined using models which included traffic density, background pollution levels, proximity to intersections, and vegetation density between the bike route and vehicle traffic as covariates.

The highest concentration (compared to background) of BC was found in the Bike lane (2360ng/m^3 (2.36μg/m^3)), for NO_2 this was found in the designated bike lane (24.2ppb).

Table 4. Exposure to NO_2 and black carbon by bike facility type

Route type	Mean (standard error)	
	Black Carbon (ng/m^3)	**NO_2 (ppb)**
Sampled concentration		
Bike path	1670 (101)	14.7 (0.582)
Bike lane	2360 (85.1)	19.5 (0.343)
Designated bike lane	1980 (336)	24.2 (1.72)
Background concentration		
Bike path	640 (16.9)	16.1 (0.115)
Bike lane	641 (9.35)	15.8 (0.102)
Designated bike lane	1020 (129)	15.9 (0.245)

In both unadjusted and adjusted models (adjusting for traffic, background concentration and intersections), bike paths had statistically significantly lower concentrations of BC and NO_2 than bike lanes. However adjusting for vegetation reduced the magnitude of the differences between bike paths and bike lanes. After adjusting for traffic density, background concentration, and proximity to intersections, bike lanes were found to have concentrations of BC and NO_2 that were approximately 33% higher than bike paths. The authors note that these variations are influenced by distance from the road, vegetation barriers, and reduced intersection density.

Table 5. P-values testing the hypothesis that the mean pollution levels for each route type are equal, adjusted and unadjusted models

Route type	BC		NO_2	
	Bike lane	**Designated bike lane**	**Bike lane**	**Designated bike lane**
Unadjusted				
Bike path	<0.0001	0.837	<0.0001	<0.0001
Bike lane	NA	0.751	NA	0.06
Adjusted for traffic, background concentration and intersections				
Bike path	0.0054	0.975	<0.0001	<0.0001
Bike lane	NA	0.663	NA	0.054
Adjusted for traffic, background concentration, intersections and vegetation density				
Bike path	0.063	0.995	0.259	0.015
Bike lane	NA	0.655	NA	0.059

Bike paths are frequently found along roads with high levels of traffic. Bike paths lead to lower concentrations than bike lanes given a set traffic density, but also have significantly lower concentrations despite higher traffic densities along the bike paths observed in this study. The authors identify two factors behind this:

- bike paths are further removed from the road than bike lanes

- physical barriers between the bike lane and motorised traffic may interfere with the dispersion of pollutants.

When vegetation is added to the model, the reduction in pollution levels that was previously attributed to the bike path is transferred to the vegetation density variable. The remainder of the difference in pollutant levels is likely to be due to the greater distance between bike paths and vehicle traffic and the reduction in intersections along bike paths. A one-unit increase in vegetation density (e.g. from sparse to moderate vegetation) was associated with a 56μg/m^3 (3.4%) reduction in BC and a 1.7ppb (11.6%) decrease in NO_2.

Boogaard (2009, -)[7] examined the quality of cycling infrastructure on a range of factors, including exposure to particulate air pollutants, in a non-randomised controlled trial. It compared exposure while cycling with exposure while driving in 11 medium sized Dutch cities.

Real time exposure to particle numbers (PNC) and $PM_{2.5}$ were measured while driving or cycling 12 predefined routes of approximately 10–20 min duration in each city. Sampling took place on eleven weekdays (excluding Fridays) between late August and October 2006 between 12:00 and 19:00. The routes were selected to give a representative picture of the infrastructure for cyclists and were within of 2.5km within the city centre. The shortest route was chosen for both modes and so were not the same for bicycle and car.

There were large variations in minute averages of PNC within cities and between cities/sampling days. The overall mean was 25,545 in the car and 24,329 particles cm^{-3} on the bicycle.

When 1-s averages were compared, typically, higher short-term peaks of PNC were measured during cycling. The 99-percentile of 1-s averages varied between 56,268 and 173,535 particles cm^{-3}; the car/bicycle ratio was 0.70 (overall 43% higher during cycling).

Higher mean PNC levels occurred in the following situations:

- Passing mopeds and to a lesser extent cars
- Passing an intersection with a right-of-way road
- Waiting for a traffic light (and when waiting for a bridge). Average waiting time for a traffic light was 25s.

Cycling on a bicycle lane (separated from other road vehicles by marking only) was associated with an increase of 11% in PNC exposure, cycling on a bike path (cyclists only, unattached but parallel with road) and a bicycle street (cyclists are main users; other road vehicles are ‘‘guests’’) showed smaller increases (8%).

Overall, the mean PNC exposure for car drivers was 5% higher than for cyclists. The authors report slightly higher 1-min peak concentrations were measured in the car (data not given). Shorter duration peaks (around 10s) of PNC were higher during cycling. A large variability of exposure was found within and between routes.

Median $PM_{2.5}$ concentrations per sampling day ranged between 5 and 112 μg m^{-3}. The overall mean was 11% higher in the car than during cycling (49.4 μg m^{-3} in the car and 44.5 μg m^{-3} on the bicycle). No significant relation was found between $PM_{2.5}$ and any of the predictor variables except for time and type of route.

The authors suggest that the lower exposure levels for cyclists may be due to their ability to dodge between vehicles or avoid major roads. Their position next to the kerb (compared to vehicle drivers who are close to the exhaust pipe although inside a vehicle) may also play a part. They note that positional factors (height of breathing zone, location of pavement or bicycle lane), traffic factors (traffic intensity, speed and composition of the selected routes; ventilations status of the car) and meteorology have been suggested as being important in determining exposure to pollutants.

Kendrick (2009[21], -) measured PNC (dominated by UFPs) in a newly installed cycle track and the site of a ‘traditional bicycle lane’ on the same road in a comparative study. The study site was a multi-lane, one-way street in the downtown Portland core. Traffic volumes and composition vary throughout the day.

Before implementation of the cycle track, the road consisted of three lanes with a ‘traditional bicycle lane’ located between the nearside travel lane and a row of kerb parking. After installation, two travel lanes remained, with an offset row of parallel parking providing a buffer to the cycle track, approximately 10 -11 feet (approx. 3m) in width. Exposure concentrations in the conventional bicycle lane versus a cycle track lane were carried out using a parked car.

UFP exposure concentrations were significantly greater on the driver’s side than the passenger’s side for all study days.

Bicycle lane exposure concentrations were always significantly greater than the cycle track exposure levels. There was a wide range in the mean of the differences and percent differences (8%-38%). Sidewalk median exposure concentration was equal to 12,900 particles/cubic centimetre (pt/cc) with a mean concentration of 15,535pt/cc and a range from 6,890-433,000pt/cc. The bicycle lane concentrations were significantly greater than the sidewalk with a mean difference equal to 6,805pt/cc, t-value=28.4, p-value<0.01. The percent difference was 38%.

Measurements on the sidewalk showed a continued significant decline in exposure concentrations from bicycle lane to cycle track to sidewalk. The authors suggest that the differences in the UFP levels are most likely due to the increased horizontal distance from the traffic stream and the airflow over the parked vehicle.

Table 6. Mean PNC, Ranges, Percent Differences, and t-test 8 Results for Bicycle Lane and Cycle Track Exposure Concentration Comparisons

		Bicycle lane			Cycle track					
Date	**Time**	**Median**	**Mean**	**Range**	**Median**	**Mean**	**Range**	**Mean diff**	**p-value**	**% diff**
Nov 24, 2009	**05:45-10:45**	31,400	43,788	14,500-500,000	30,500	37,498	15,000-365,000	6,125	<0.01	15
Nov 24, 2009	**10:58 - 13:52**	28,200	56,845	4,510-500,000	26,000	35,802	13,600-500,000	21,043	<0.01	38
Nov 24, 2009	**14:05-16:51**	25,400	37,476	9,980-500,000	20,600	24,618	2,230-312,000	12,589	<0.01	35
Feb 8, 2010	**05:31-10:49**	30,600	47,601	12,300-500,000	29,500	44,245	3,340-500,000	3,309	<0.01	8
June 7, 2010	**06:53-14:20**	14,700	25,271	3,340-500,00	14,200	20,805	5,750-500,000	4,465	<0.01	18
July 13, 2010	**07:24 - 21:42**	8,290	13,839	2,390-500,000	7,660	10,558	5,620-500,000	3,309	<0.01	24

Overall summary

The studies identified show that siting of cycle lanes can have an impact on cyclists' exposure to poor air quality. Key features reducing poor air quality are avoiding the use of busy roads, distance from motor vehicles, the use of foliage as a screen and avoiding unnecessary delays in areas of high pollution (such as busy junctions).

Evidence statement 3.1 features of cycle lanes affecting exposure to air pollution

There was inconsistent evidence from 6 [-] studies (3 from the US[20],[23], [21], one from the UK[6], one from the Netherlands[7] and one from Canada[19]) showing that siting and design of cycle routes can influence cyclists' exposure to air pollution. Increasing the separation between cyclists and motor vehicles, or using foliage to screen lanes reduces exposure to vehicle related air pollutants (UFP, PNC, black carbon and NO_2); exposure to $PM_{2.5}$ increased by increasing separation.

One study examined differences in lung function following use of a high traffic or a low traffic cycle route. All changes in lung function were not significant[20].

Two studies[20,19] found exposure to $PM_{2.5}$ was lower in the high-traffic route; this was not statistically significant. Both found exposures to other traffic related pollutants (black carbon and UFPs) to be raised, in 1 case exposure to UFPs reached significance[20] (increase in exposure from 13,500 to 19,950 particles/cm^3). One study[7] found exposure of drivers to $PM_{2.5}$ to be 11% higher than for cyclists.

All six studies found exposure to other traffic related air pollutants (BC, UFP, NO_2) was higher in the high-traffic route, in some cases this reached significance.

Black carbon was found to be 33% higher in bike lanes than paths[23] after adjusting for traffic density, background concentration and junctions; each 5m separation reduced levels by 2.5% (95% CI: -17, 12)[19]; Jarjour[20] found mean BC levels increased from 1.76μg/m3 to 2.06μg/m3 from low to high traffic routes.

UFP reductions from 19,945.34 to 13,517.13 particles/cm^3 were found in one study[20], and an increase in distance between cycle routes and motor vehicles of 5m was found to reduce PNC by 1.2% (95% CI: -10, 8.5)[19]. Moving a cycle facility to the nearside of a parking lane reduced PNC by 25% ($p<0.01$). In 1 study[7] the mean level of PNC for drivers was 5% higher than for cyclists. Peaks occurred in conjunction with passing vehicles (especially mopeds), crossing right of way roads and waiting at signals.

NO_2 reductions of between 37% and 41%, with a mean of 37.5%, were found in 1 study[6] ($t = 11.78$; $P < 0.01$) which looked at compared journeys using 'on road' and 'off road' routes. Exposure taking into account time taken to cycle the route was also significantly reduced ($t = 3.50$; $P = 0.017$).

One study which examined the impact of vegetation[23] found that a one-unit increase in vegetation density (e.g. from sparse to moderate vegetation) was associated with a 56μg/m3 (3.4%) reduction in BC and a 1.7ppb (11.6%) decrease in NO_2

Applicability: one study is from England and so applicable. Other studies come from the US, Canada or the Netherlands; differences in vehicle fleets, city form and meteorology make this evidence partially applicable.

20 Jarjour et al. 2013 (−)

23 MacNaughton et al. 2014 (−)

6 Bean et al. 201 (−)

19 Hatzopoulou et al. 2013 (−)

7 Boogaard et al. 2012 (−)

21 Kendrick et al. 2009 (−)

4.2.2 Public transport

Gramsch et al (2013[17], -) investigated the effect of a change in the public transportation system and bus fleet (Transantiago) on levels of black carbon air pollution in Santiago, Chile in a controlled before and after study.

Before Transantiago, the city had a fleet of about 7000 diesel buses operated by a large number (around 3,000) of bus owners. After implementation (which was carried out in a single phase) the bus fleet was reduced to about 5900, of which about 1500 were new Euro III-standard buses. All Euro I buses were taken out of circulation. Measurements of black carbon were taken before (June-July 2005) and after (June-July 2007) the intervention along 4 roads (3 crossing the city with main avenues directly affected by the intervention - Usach, Alameda and Departamental) and 1 where no public transportation was available before or after the intervention - E. Yañez. Alameda is described as a ‘trunk bus’ route; all buses on Alameda were Euro III compliant. Usach and Departamental were mixed with a combination of Euro III and buses with over 3 years of service.

Table 7. Number of buses and other vehicles per day, 2005 and 2007 for all sites

	2005			2007		
	Buses	Other diesel	Total	Buses	Other diesel	Total
Usach	361	216	5606	247	230	5940
Departamental	1509	5580	36475	825	5507	35975
Alameda	6730	5422	77370	4350	3767	53760
E Yañez		2315	32398		2480	34753

The only site which showed a decrease in average BC levels (19.31μg/m^3 – 11.93μg/m^3) after the intervention was Alameda street. No other sampling sites showed a decrease in average levels.

Table 8. BC levels before and after Transantiago.

Site	**Year**	**Sampling period**	**BC average ± s.d. (μg m^{-3})**	**Min (μg m^{-3})**	**Max (μg m^{-3})**	**number of measurements**
Usach	2005	1 June – 29 July	7.91 ± 5.69	0.00	33.55	1303
	2007	1 June – 31 July	8.29 ± 5.78	0.05	47.02	1437
Alameda	2005	1 June – 4 July	19.31 ± 9.50	0.64	59.68	801
	2007	6 June – 31 July	11.93 ± 7.64	0.40	59.80	1317
Departamental	2005	2 June – 2 July	9.36 ± 5.67	0.00	26.71	715
	2007	4 June – 31 July	10.21 ± 7.93	0.00	124.65	1389
E. Yañez	2005	1 June – 2 July	5.05 ± 2.87	0.03	19.76	753
	2007	29 June – 31 July	5.93 ± 3.81	0.16	23.71	483

The authors note that the lack of a drop in BC at streets other than Alameda may have been because the drop in bus numbers is offset by an increase in the number of other vehicles or in the number of other diesel vehicles (diesels on Usach fell from 577 to 477 while total number of vehicles increased from 5606 to 5940; on

Departamental although there was a drop to 35,975 VPD from 36,475 total number of diesels increased from 7089 to 8117. At Yañez both total number of vehicles and number of diesels increased). The authors suggest the increase in numbers of vehicles was a response to the reduction in overall bus service and confusion about new routes. There was no increase in total traffic at Alameda as the road was already at full capacity.

Burgard et al (2009, -)[9] used a remote sensing device to measure on-road and in-use gaseous emission measurements from three fleets of schools buses in a non-randomised controlled trial. The fleets were one unmodified fleet (at Vashon Island) and two modified fleets at two locations in Washington State using either diesel oxidation catalyst (DOC) and diesel particulate filter (DPF) equipped buses. The control fleet was on Vashon Island. This fleet had the most comparable engines to the Bainbridge Island fleet but had not been retrofitted with soot-reducing devices. Emissions from buses were measured in the morning, loaded with students before entering the school drive, and then pulling away after dropping off students. In the afternoon, emissions were first measured from empty buses pulling into the school and then with them loaded with children when leaving.

The sensors were sited near the school gates. Measurements were taken between 21 May and 9 October 2007.

The retrofit technologies resulted in an increase in emitted NO NO_2 and NO_x. The authors note the differences in the age of the 3 vehicle fleets.

Table 9. NO_2 emissions – retrofit and control fleets

	Retrofit fleet average (g/kg)		Control fleet average (g/kg)
	DPF (n=74)	DOC (n=53)	All buses (n=162)
NO_2*	17.2 ± 4.5	4.4 ± 1.1	3.8 ± 0.8
Model year	2000	1993 - 1995	1995 - 2002

Evidence statement 3.2 changes to bus services and technology

Inconsistent evidence from 2 [-] quality studies, 1 from Chile[17] and 1 from the US[2] shows mixed effects from changes to bus emission control standards and services.

Substantial changes to the organisation and improvements to the emission control standards of public transport buses in Santiago[17] showed an effect on black carbon (a reduction from 19.31 ± 9.50 to 11.93 ± 7.64μg/m^3) only where there was a large drop in traffic as well as a reduction in bus numbers and improvements in technology.

Retrofit of diesel oxidation catalyst or diesel particulate filter technologies on school buses was associated with higher NO_2 emissions compared to a control fleet without retrofitted technology (3.8±0.8 g/kg fuel used (control fleet) to 4.4±1.1 (DOC) and 17.2±4.5 (DPF))[9].

Applicability: The studies are from Chile and the US. Differences in the bus fleet make up and in technologies and emission standards means that this evidence is of limited applicability.

17 Gramsch et al. 2013 (−)

9 Burgard et al. 2009 (−)

4.2.3 Use of alternative fuels

Soret (2014[26], -) modelled the degree of vehicle electrification (low, medium and high levels) required to reduce the present air quality problems in Barcelona and Madrid. The air quality impacts of fleet electrification were analysed for a critical episode of air pollution affecting the entire Iberian Peninsula (3-5 October, 2011), characterised by a high pressure system situated over the Iberian Peninsula.

The base case scenario was the current situation in 2011 with no fleet electrification. The three scenarios were:

- Low: ~13 % electrification.
- Medium: ~26% electrification.
- High: ~40% electrification.

The study was performed by applying the WRF-ARW/ HERMESv2/ CMA Q model[3] system at high spatial (1 x 1 km^2) and temporal (1h) resolution.

The total vehicle-kilometres-travelled (VKT) are estimated as13,462,321 and 25,787,145 in Barcelona and Madrid respectively. In the scenarios, electric and hybrid drive modes replace a percentage of the current vehicles, maintaining a constant ratio of the current categories of conventional vehicles instead of replacing only old vehicles (greatest polluters).

The estimated energy demand for EV charging (High scenario) represents ~0.4% of total daily energy demand in Spain.

3 http://www.wrf-model.org/index.php

The analysis is focused on the differences between the Base case and High (~40% fleet electrification) scenarios. However, the differences of the Low (~13%) and Medium (~26%) scenarios are proportional. In terms of emissions, the high scenario (~40%) showed reductions of 11% and 17% of the total NO_x emissions in the cities of Barcelona and Madrid, respectively. These emission reductions involve air quality improvements in NO_2 of 8-16% for the maximum hourly values: reductions up to 30 and 35 mg m^{-3} in Barcelona and Madrid, respectively.

NO_x traffic reductions (between the Base case and High scenario) are similar in both cities: 27 and 25% in Barcelona and Madrid, respectively. The differences are related to the vehicular fleet composition of each city: LDVs are more common in Barcelona, while Madrid has a higher percentage of HDVs. However, the total NO_x reduction is lower in Barcelona (11%, 31.42-28.06mg/day) than in Madrid (17%) 41.27-34.13mg/day).

Fleet electrification has a limited impact on PM_{10} reductions in terms of both emissions (5.89-5.71mg/day (Madrid), (3.06-2.95mg/day (Barcelona) (3-4%) and air quality (2-5%)). This is due to the high contribution of non-exhaust emissions (resuspension and brake, tyre and road abrasion), which are not reduced by electrification.

Somewhat higher reductions of $PM_{2.5}$ emissions (2.5-2.37mg/day, Barcelona and 4.66-4.45mg/day, Madrid) are observed because exhaust road transport emits the fine fraction of PM. These higher reductions lead to slightly higher air quality improvements (3-7%) in Barcelona and Madrid respectively.

Although fleet electrification leads to significant improvements in emissions of pollutants, the reductions achieved do not prevent episodes of poor air quality. The authors note that this suggests that electrification cannot be considered the sole solution, especially regarding particulate matter. The percentage of fleet electrification examined (26-40%) is substantial, suggesting that fleet electrification will have a limited impact on air quality for at least the next decade. They note that fleet electrification and other management measures should involve all vehicle categories (two-wheelers, heavy-duty vehicles, buses and light-duty vehicles) not only passenger cars.

Goncalves (2009a[15], -) modelled seven scenarios with different levels of conversion of the vehicle fleet to natural gas in Madrid and Barcelona.

The study used the high resolution (1km^2, 1 hr) HERMES emissions model[4] specific to the Iberian peninsula. The scenarios examined were:

- E1: transformation to NGV (natural gas vehicles) of 1005 of urban bus fleet

[4] https://www.bsc.es/earth-sciences/hermes-emission-model

- E2: transformation of 50% of taxi fleet
- E3: transformation of 50% of intercity bus fleet
- E4: transformation of 50% of light commercial vehicle fleet
- E5: transformation of 10% of private car fleet
- E6: transformation of 100% of heavy duty freight fleet
- E7 scenarios E1-E6 combined

There is a larger overall vehicle fleet in Madrid than Barcelona (1.7million compared to 1m vehicles), while economic activity in Madrid is dominated by the service sector and in Barcelona is predominantly industrial. Thus contribution to NO_x emissions from road traffic is higher in Madrid than in Barcelona (94% and 81% respectively). Industrial contribution to PM_{10} accounts for 35% of the total in Barcelona and around 2% in Madrid.

The overall combined scenario reduced $PM_{2.5}$ emissions by 40% and 46%, PM_{10} by 41% and 34% (Barcelona and Madrid respectively) and NO_x by 27% and 35% (only percentage changes from base case reported). There was a difference in the most effective scenario for reducing NO_x emissions in the two cities. The largest effect in Barcelona came from changing light commercial vehicles (E4), while in Madrid it came from changing 10% of private cars.

Table 10. Change in NO_x and PM_{10} emissions for various scenarios of substitution with natural gas vehicles, Barcelona and Madrid

	Barcelona		Madrid	
	NOx (kg d-1), % change	PM_{10} (kg d-1), %change	NOx (kg d-1), % change	PM_{10} (kg d-1), %change
Base case	23,949	7,356	66,700	18,238
E1	-3.6%	-3.1%	-2.7%	-2.9%
E2	-2.8%	-4.2%	-1.8%	-3.9%
E3	-2.0%	-1.8%	-3.8%	-4.3%
E4	-15.1%	-24.5%	-6.7%	-13.9%
E5	-7.8%	-4.6%	-10.9%	-8.1%
E6	-3.4%	-2.8%	-1.3%	-1.3%
E7	-34.7%	-41.0%	-27.3%	-34.3%

The authors note that substitution with natural gas vehicles needs to reach a critical value (around 4%) to be effective in reducing emissions.

A second modelling paper (**Goncalves 2009b[16], -**) looked at the air quality impacts of the same change scenarios. Modelling is carried out using the WRF-ARW/HERMES/CMAQ modelling system with high spatial–temporal resolution (1km^2, 1h).

The modelling was based on typical of the summertime situation in southwestern Europe. These conditions dominate annual and the summertime transport patterns. They are frequently associated with local-to-regional episodes of air pollution.

Table 11. changes in 24-hour average NO_2 and PM_{10} levels, Barcelona and Madrid for base case and 7 natural gas substitution scenarios

	Barcelona area						Madrid area					
	NO_2 24-h average			PM_{10} 24-h average			NO_2 24-h average			PM_{10} 24-h average		
	Conc (µgm-3)	Change		Conc (µgm-3)	Change		Conc (µgm-3)	Variation (%)		Conc (µgm-3)	Variation (%)	
		Δ conc	%		Δ conc	%		Δ conc	%		Δ conc	%
Base case	35.0			10.4			22.2			4.9		
E1	34.8	-0.20	-0.56%	10.4	-0.05	-0.48%	21.8	-0.44	-1.98%	4.9	-0.06	-1.21%
E2	34.9	-0.15	-0.15%	10.4	-0.07	-0.66%	22.0	-0.28	-1.24%	4.8	-0.08	-1.64%
E3	34.9	-0.11	-0.11%	10.4	-0.03	-0.29%	21.6	-0.61	-2.73%	4.8	-0.09	-1.79%
E4	34.1	-0.89	-0.89%	10.0	-0.42	-3.99%	21.1	-1.12	-5.04%	4.6	-0.31	-6.24%
E5	34.6	-0.46	-0.46%	1034	-0.08	-0.74%	20.4	-1.82	-8.2%	4.7	-0.17	-3.49%
E6	34.8	-0.19	-0.29%	10.4	-0.05	-0.44%	22.0	-0.21	-0.96%	4.9	-0.08	-0.54%
E7	32.9	-2.15	-2.15%	9.7	-0.69	-6.6%	17.7	-4.58	-20.56%	4.2	-0.73	-14.92%

The largest reductions in NO_2 and PM_{10} pollution are achieved by the combined scenario. The largest individual contribution comes from a substitution of 50% of light commercial vehicles with NGV (scenario E4) and a substitution of 10% of private cars (scenario E5). This is particularly significant in terms of reductions of NO_2 in Madrid (8.2% (1.82µg/m^3) reduction in 24-hour average levels). Changes in air quality in Madrid are greater than in Barcelona because traffic sources are a larger percentage contributor in Madrid than in Barcelona. Conversely, NO_2 and PM_{10} levels are higher in Barcelona.

Chong (2014, +)[11] modelled the air quality implications of adopting alternative propulsion technology (lean-burn compressed natural gas, hybrid electric buses) and emissions control strategies (continuously regenerating trap, exhaust gas recirculation and selective catalytic reduction with trap) in the diesel bus fleet in Greater London.

The paper uses a bus traffic model created to spatially simulate the Greater London bus network. Four scenarios (labelled SCRT, EGRD, CNGL and HYBR) were defined, as well as the current baseline. The baseline represented the 2010 composition of 8624 Euro-II[5] to Euro V buses with particle filters to meet Euro IV PM limits. The make-up of the other scenarios are set out in the table below.

Table 12. Definition of bus scenarios by percentage of drivetrain and emissions control strategies

Emission control standard and technology	Scenario name				
	Baseline	SCRT	EGRD	CNGL	HYBR
Euro-II +CRT	23.9%	-			
Euro-II +SCR +CRT	-	23.9%			
Euro-II +EGR +DPF	-	-	23.9%		
Euro-III +CRT	48.5%	-			
Euro-III +SCR +CRT	-	48.5%			
Euro-III +EGR +DPF	-	-	48.5%		
Euro-IV +SCR +CRT	15.9%	15.9%	15.9%		
Euro-IV +EGR +DPF	3.6%	3.6%	3.6%		
Euro-V +SCR +CRT	6.7%	6.7%	6.7%		
Euro-V +EGR +DPF	1.4%	1.4%	1.4%		
Lean burn CNG	-			100%	
Hybrid electric diesel	-				100%

Emission control technologies: CRT = continuously regenerating trap; SCR = selective catalytic reduction; EGR = exhaust gas recirculation; DPF = diesel particulate filter.

Air quality modelling using WRF weather forecasting and the regional chemistry-transport model CMAQ[6] were used to simulate one month per season: January, April, July and October. Health impact changes were modelled using the population aged 30 and older. A concentration response function of 1% (range 0.4-1.8%) decrease in all-cause deaths per µg/m^3 in annual $PM_{2.5}$ exposure was used.

The population-weighted $PM_{2.5}$ concentration decrease from the BASE to the SCRT, EGRD, CNGL, and HYBR scenarios by 36%, 14%, 63%, and 37%, respectively.

The model estimates that BASE scenario bus emissions cause 5 deaths (90%, CI 2 to 8) deaths in Greater London annually, around 1% of the premature mortalities in Greater London due to the emissions from road transport. Estimates show that greater than 89% of the bus emissions-attributable premature mortalities were avoided in the CNGL scenario. SCRT and HYBR scenarios cause 4 (90%, CI 2 to 7) and 4 (90%, CI 4 to 6) premature mortalities a year in Greater London respectively. No health benefit from the EGDR scenario compared to BASE was found.

Summary

[5] Euro standards are EU defined emission standards for vehicles. http://ec.europa.eu/environment/air/transport/road.htm

[6] https://www.epa.gov/air-research/community-multi-scale-air-quality-cmaq-modeling-system-air-quality-management

Four modelling studies looked at the impact of changing the fuel use by the vehicle fleet. They considered Madrid, Barcelona and London. Larger reductions are seen with increasing electrification of the fleets, in particular on emissions of NO_2. Air quality lags behind this somewhat, as other sources of pollutants are unaffected by changes to the vehicle fleet.

Modelling evidence statement 3.3 Use of alternative fuels

Weak evidence from 3 [-] quality studies from Spain[26, 15, 16] and 1 [+] from the UK[11] show considerable potential improvements in air pollution emissions from changes to the fuel used on air pollutants, assuming the penetration of these technologies is large enough.

Fleet electrification (~40%) showed reductions of 11% and 17% of the total NO_x emissions in Barcelona and Madrid, respectively and air quality improvements in NO_2 of 8-16% for the maximum hourly values: reductions up to 30 and 35 mg m-3 in Barcelona and Madrid, respectively. In contrast, it has a limited impact on PM_{10} reductions in terms of both emissions (3-4%) and air quality (2-5%). Slightly higher reductions in air quality improvements are seen for $PM_{2.5}$ (3% and 7% in Barcelona and Madrid)[26]

Conversion of the vehicle fleet to natural gas in Madrid and Barcelona needs to reach a critical value (around 4%) to be effective in reducing emissions (NO_2, $PM_{2.5}$)[15,16].

The largest individual contributions come from a substitution of 50% of light commercial vehicles with natural gas vehicles (in Barcelona) and a substitution of 10% of private cars (in Madrid). This is particularly significant in terms of reductions of NO_2 in Madrid (8.2% reduction in 24-hour average levels). Changes in Madrid are greater than in Barcelona because traffic sources are a larger percentage contributor in Madrid than in Barcelona. Conversely, pollutant levels are higher in Barcelona.

Estimates for London show a modelled reduction of 63% in population weighted $PM_{2.5}$ exposure and that more than 89% of the bus emissions-attributable premature mortalities were avoided by conversion of 100% of buses in London to lean burn compressed natural gas[11].

Applicability. These studies were carried out in Spain and the UK. Studies are partially applicable as results are site specific and there may be differences in the vehicle fleet make up between Spain and London and the rest of the UK.

26 Soret et al. 2014 (−)

15 Goncalves et al. 2009a (−)

16 Goncalves et al 2009b (−)

11 Chong, 2014 (−)

4.2.4 Use of alternative fuels: Cost effectiveness studies

Cohen et al (2003)[13] [+] examined the incremental cost effectiveness of buses using emission controlled diesel(ECD), (continuously regenerating particulate filters with low-sulphur fuel) compressed natural gas (CNG) and conventional diesel (CD). The paper examines the resource costs and health benefits associated with the purchase of a fleet of new ECD or CNG buses compared with a fleet of new CD buses. It aims to provide an indication of the likely answer, the key sources of uncertainty and to provide a framework for a more detailed analysis.

The paper looks at a hypothetical transit district and quantifies the health impacts from exposure to ozone and PM. This expresses atmospheric modelling results as the fraction of pollutants or their precursors emitted from a source eventually inhaled by some member of the population.

Vehicle operation emissions and emissions from upstream activities (feedstock extraction and fuel production operations) were considered.

Intake fractions for primary PM (particles emitted directly from the vehicle), secondary PM (produced by the processes of atmospheric chemistry) and ozone are used to calculate population exposure, and the intake fraction for primary PM is divided into near-source and far-source contributions. Contribution of NO_x and SO_2 to secondary PM is estimated, as is the amount of ozone inhaled per unit of NO_x emitted. An average value across six geographic regions is used as the central estimate to account for volatile organic compounds (VOC) and ultra violet light (UV) concentrations. Production of secondary PM occurs through complex chemical reactions in which VOCs and UV are involved.

Values for the contribution of ozone and PM to all-cause mortality from the literature are used to estimate life years lost per 1m population exposed per µg. The authors make the assumption that victims suffered from pre-existing coronary or respiratory disease and that each life year is worth 0.8 QALYs (source for this assumption not stated). Estimates for lung cancer from either diesel or CNG exhaust are also included. Due to the uncertainty around this link the lower bound estimate used is zero.

Greenhouse gas emissions from upstream and vehicle operation are included using estimates of the incremental damage or optimal carbon tax calculated using integrated assessment models. Transit agency costs (vehicle procurement, infrastructure and operations) are included.

Overall, the authors report that purchasing a fleet of 1,000 ECD buses saves 6 QALYs a year relative to a fleet of CD buses. For CNG buses the figure is almost 9. Omitting monetized impacts of greenhouse gases gives a central cost effectiveness ratio (calculated by dividing the central estimate values for incremental QALYs saved into the central estimate of financial costs) of $270,000 for ECD and $1.7-2.4million (low and high cost region – low and high cost refer to costs associated with infrastructure costs relating to fuelling and maintenance) for CNG. Relative to ECD, CNG costs $5-8million per QALY saved. Inclusion of monetized greenhouse gas costs reduces these estimates by around 5% using high end estimates. There is considerable uncertainty in these estimates based on the upper and lower bounds reported. The range of QALYs saved for ECD is between 0 and 41, and for CNG is 0.01 to 65. This gives a CE ratio for ECD of between $30,000 and infinite (no heath gain). For low cost areas, the range of CNG is between $70,000 and $2billion.

The authors note that the ICERs are higher than for interventions in a range of other areas such as medical prevention of cancer and heart disease, reduction of motor-vehicle trauma and prevention of infectious diseases. However, costs per unit health benefit from different government agencies varies substantially. Median costs per life year saved by regulations considered by different agencies varied from $7.6million (Environmental Protection Agency), $23,000 (Federal Aviation Authority), $68,000 (Consumer Product Safety Commission), $78,000 (National Highway Traffic Safety Administration) and $88,000 (Occupation Safety and Health Administration). Comparing cost effectiveness ratios in the most relevant context (reducing human exposure to air pollutants) shows that stationary source control (i.e. sources of air pollution that do not move) CE ratios are in the order of 10s of thousands to more than $1m per QALY. For mobile sources (such as vehicles), the range is from 10s of thousands to $4m. Thus the CE for ECD falls within the middle of the range and for CNG towards the higher end for mobile sources.

A number of simplifying assumptions in the analysis are identified. These include the emissions data which is limited to a relatively small set of dynamometer measurements. These do not identify how maintenance may impact on emission levels. Secondly, exposure estimates do not take account of atmospheric complexities or specific characteristics. The exposure metric assumes that annual average exposure is the proper dose metric and does not allow for dose-rate dependence or thresholds. Thirdly, the possibility that the impact of PM is due to UFP is not considered. If the impact of particulate matter is due to the ultra-fine fraction and these are not proportional to $PM_{2.5}$ the health impact may vary. If this were to be the case the results may be invalid.

The authors also note that there are a number of other factors which may be important. These might include odour and safety risks from new fuels.

Cohen (2005)[12] [+] uses the same approach and assumptions to examine the cost effectiveness of changes to school bus fleets from conventional diesel (CD) to ECD or CNG.

Three hypothetical school districts are considered. These differ in terms of their size, population density and distance travelled by each bus annually. They are intended to be roughly representative of the range of conditions encountered in the US. Two of the scenarios are 'large systems' based on the characteristics of the 100 largest school bus fleets. The scenarios differ in population density and so the cost of acquiring land for new infrastructure varies. One scenario, described as 'dense urban' assumes heavy land use making acquisition expensive (population density 1600/km2). The second, referred to as 'moderate urban' assumes that acquisition of land for infrastructure is 'inexpensive' (population density 400/km2). The third scenario represents a small system (population density 26/km2). [Population density of Metropolitan Boroughs such as Wigan is around 1600/km2, while Kettering and Mid Sussex have densities of around 400/km2].

For the dense urban scenario, ECD reduces annual health damages by 3.8 QALYs annually and CNG reduces them by 3.9 QALYs annually.

Table 13 central estimate cost-effectiveness ratios

Scenario	**Cost effectiveness of ECD ($/QALY)**	**Cost effectiveness of CNG ($/QALY)**
Dense urban	450,000	4,200,000
Moderate urban	640,000	3,600,000
Small system	900,000	4,000,000

The author identifies key factors contributing to uncertainty as 1) the intake fraction for inhaled PM derived from far-source PM emissions; 2) the health damage associated with exposure to PM. Resource cost also contributes to the uncertainty for CNG but not ECD for which there is less infrastructure additional cost as the infrastructure for ECD is closely related to the current infrastructure whereas for CNG considerable changes would be required. The impact of the carcinogenicity of diesel exhaust and incorporation of greenhouse gas damage costs do not contribute substantially to the uncertainty of the results.

The author notes the caveats associated with both studies – that the analysis relies on limited emissions data; that the models do not take into account atmospheric complexities; that the health risks of PM may be associated with UFPs. The analysis also focuses on aggregate population risk and so individuals living near bus routes will be exposed to substantially greater risk.

A paper by **Krutilla and Graham (2012)[22] [+]** examines the Net Present Value time paths of pick-up and delivery (PUAD) diesel hybrid vehicles. NPVs are looked at from five perspectives: transportation firms, parties impacted by externalities, state and local governments and wider society (the sum of the other stakeholder effects).

The principal trade off evaluated is the difference between the societal value of diesel fuel savings and the incremental costs of diesel-electric hybrid vehicles.

Factors taken into account include:

- Fuel savings (the pre-tax (diesel) price represents the societal value of diesel fuel savings)
- Externalities related to fuel consumption – CO_2 emissions, national security risks linked to oil imports
- Externalities linked to tailpipe emissions
- Externalities linked to vehicle miles travelled (VMT) – congestion, injuries, road maintenance. These are not directly included in the calculations as VMT are likely to be similar.
- Technology costs
    - Additional capital cost
    - Additional costs of battery replacement
    - Differences in operation and maintenance costs

Model parameters include hybrid performance and fuel savings, diesel fuel price and taxes, global climate benefits, energy security, air quality benefit and time saved in refuelling. Cost side elements include costs of diesel-electric hybrids, battery costs and taxes. Under the assumptions, taxes are transfers and will not affect the societal returns of hybrid utilization. They will have a significant effect on private returns and state and federal revenues. Taxes include fuel, sales and corporate income tax.

Various discounting rates are used based on different perspectives. Societal discount rate of 3% and 7% are used. Private discount rate of 7% is used.

NPV for each stakeholder group for a single year and across the simulation horizon is examined. Stream of NPVs then discounted back to the present, showing result of treating the temporal profile of yearly NPVs as an investment option for policymakers to incentivize.

Societal returns require depend on voluntary behaviour of commercial firms so private incentives need to be congruent with socially desirable outcomes for social benefits to be realized. A stakeholder assessment gives insights about the degree of financial transfer required to incentivize private firms to purchase new technology.

The first scenario uses a 7% discount rate for both private and societal costs. Summing all effects on the private hybrid purchaser yields an expected NPV loss in 2024 of -$8450. The net society perspective is -$6120.

Using a 3% social discount rate and a 7% private discount rate alters the overall NPV. Under this scenario, the expected NPV is positive, at $2323, justifying the investment from a societal perspective.

The paper presents temporal simulation paths. These demonstrate that the introduction of fuel consumption standards have reduced the NPV. The expected value of societal returns to hybrid investment generally start out negative and become positive in the longer run.

Under scenarios with high values for technology cost, fuel economy and fuel prices (HHH) the expected NPV in 2017 is about $1196 compared with over $17,000 without the changes in fuel standards. With the societal discount rate at 7% the expected NPV becomes positive around 2020, reaching around £20,000 by around 2030. For the private investor it becomes positive under the HHH scenario around 2024 but never becomes positive for the low values for technology cost, fuel economy and fuel prices (LLL) scenario.

They note that the magnitude of net benefits depends on the voluntary usage of energy technology and that tax distortions, discount rates and regulations that influence behaviour should be reflected in BCA. With the exception of fuel taxes, tax distortions are largely ignored in analyses of fuel saving measures in the commercial sector. Correctly specifying tax status is a necessary condition for understanding private perspectives on energy investments. That knowledge is necessary for considering financial incentives required to induce private firms to undertake socially desirable actions.

Evidence statement 3.4: cost-effectiveness of changes to vehicle fleets in relation to fuel type

Three [+] examined the cost effectiveness of changes to vehicle fleets from the reduction in air pollution. Two studies[12, 13] showed that fleet changes are not cost effective approaches to reducing outdoor air pollution when compared to medical or other public health interventions. Changes to urban or school transport from conventional diesel (CD) to either emission controlled diesel (ECD)[13,12]; or compressed natural gas (CNG)[13,12] whilst having estimated health benefits resulting in reductions in QALYs losses are not considered cost effective. ECD vs CD: Urban Transport buses ($270,000 per QALY) and School buses ($400,000– 900,000 cost per QALY saved). *CNG vs CD*: Urban Transport buses ($270,000 per QALY) and School Buses (around $4million per QALY saved).

The CE ratios identified are within the range normally considered cost effective by other US government agencies for interventions that address mobile or stationary sources of air pollution.

Switching to hybrid diesel-electric fleets for urban vehicles was also judged to have a net present value below that considered to be justifiable[22], however these calculations are highly dependent on discount rates used and on perspectives.

Applicability: all studies were undertaken in the USA.

13 Cohen 2003 (+)

12 Cohen 2005 (+)

22 Krutilla and Graham 2012 (+)

4.2.5 Construction of a bypass

Burr et al (2004, -)[10] examined whether respiratory health improves in residents exposed to traffic congestion and pollution following a reduction in exposure to traffic related air pollutants related to the opening of a by-pass in a controlled before and after study.

A respiratory survey was conducted among the residents of an area in North Wales where houses flanked a main road (‘congested’ streets(n=165)). For comparison, this was also carried out in nearby ‘uncongested’ streets(n=283). The survey was conducted at baseline and again a year after the by-pass opened. Peak expiratory flow was also measured.

When the by-pass opened, the volume of heavy goods traffic fell by nearly 50%. PM_{10} decreased by 23% (8.0 mg/m^3) in the congested streets and by 29% (3.4 mg/m^3) in the uncongested streets, with similar proportionate falls in $PM_{2.5}$.

There were no clear or consistent differences between the residents of the two areas initially in terms of symptoms or peak flow variability. Repeat questionnaires were obtained from 165 and 283 subjects in the congested and uncongested areas respectively, and showed a tendency for most symptoms to improve in both areas. For chest symptoms, the improvement tended to be greater in the uncongested area, although the difference between the areas was not statistically significant. For any wheeze symptoms, the net improvement in congested streets was 0.6% and 7.1% in uncongested streets (difference in net% better −6.5%, 95% CI −14.9 to 2.0).

Symptoms relating to the nose and eyes tended to improve more in the congested streets. 'Any rhinitis' symptoms improved by 11.2% in congested streets, compared with 6.4% in uncongested streets (difference 5.4%, 95% CI −3.4 to 15.0). For 'rhinitis interfering with daily activities' the was significant (10.3%, 95% CI 3.1% to 17.3%) although the authors note this could be a chance finding due to the number of associations being examined.

Coefficient of variation of peak flow tended to increases slightly in congested streets (5.09 – 5.32, +0.23 (morning) and 5.09 – 5.25, +0.16 (evening)) and to decrease in uncongested streets 6.17 – 4.99 (−1.18 (morning) and 5.77 – 5.22, −0.55 (evening)).

The authors note that many of the subjects who participated moved away during the study, and it is possible that there was selective bias in those who moved. They note however that there was no evidence of a difference between the congested and uncongested streets in numbers moving during the study.

Evidence statement 3.5 Bypass construction

Weak evidence from one [-] study from the UK[10] found reductions in particulate matter following the opening of a bypass. PM_{10} levels fell from 35.2 to 27.2μg/m^3 (22.7%) in congested streets and from 11.6 to 8.2μg/m^3 (28.9%) in uncongested streets, with similar proportionate falls in $PM_{2.5}$. Respiratory symptoms showed a tendency for most to improve in both areas. For chest symptoms, the improvement tended to be greater in the uncongested area, although the difference between the areas was not statistically significant. Symptoms relating to the nose and eyes tended to improve more in the congested streets. For 'rhinitis interfering with daily activities' this was significant although the authors note this could be a chance finding due to the number of associations being examined. Coefficient of variation of peak flow tended to increases slightly in congested streets (+0.23 (morning) and +0.16 (evening)) and to decrease in uncongested streets (−1.18 (morning) and −0.55 (evening))

Applicability: the study is from the UK. However, the bypass opened in 1998 so it is likely that there will be substantial differences in the vehicle fleet. It is may therefore only be partially applicable

10. Burr et al. 2004 (−)

4.3 Review question 4: Are measures to promote absorption, adsorption or impingement deposition, and catalytic action

effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

Ten studies were included in the review. Overall, the quality of the studies was poor with 7 studies graded [-] and 3 studies graded [+] (see Table 3.2.3).

Studies were grouped by the intervention the study assessed:

- Use of natural and artificial barriers (5 studies)
- Use of dust suppressants (2 studies)
- Street washing (1 study)
- Urban greening (3 studies)

4.3.1 Use of natural and artificial barriers

Barriers (whether artificial such as noise barriers or natural such as trees) have the potential to affect transport related air quality in a variety of ways. Solid barriers may change in the way a plume of polluted air moves and so comes into contact with people. Stands of trees, as well as altering air flow, may increasing the deposition of pollutants through the physical impaction of materials on the plant leaves.

Al-Dabbous (2014, -)[3] carried out monitoring of PNCs adjacent to the A3 in Guilford, UK. The A3 has four lanes and carries around 6000 – 6600 vehicles per hour.

A 2.2m wide vegetation barrier 0.3m from the east side of the road separates the road from a busy footpath approx. 2m wide and 1.45m below road level. The vegetation was made up of coniferous plants providing a densely foliated tree line. The authors note openings were only provided by the space between the tree leaves and the branches. The full height of the barrier between crown and the bottom stem near the ground level was covered in green leaves. No measurement of the density of the leaves (e.g. leaf area index) is given.

Four sites were used to measure PNC concentrations: L1 was in a vegetation-free point parallel to the front of the barrier; L2 was parallel to L1 at the front of the vegetation; L3 and L4 were in the middle and back of the barrier respectively. Measuring sites were approximately 1.6m above the footpath ground level and 0.3m above street level.

The total PNCs at the sampling locations L2, L3 and L4 were found to decrease gradually with the increasing distance from the edge of the road through the vegetation barrier.

PNCs at L2 ($1.99\pm 1.77\text{x}10^5$ cm^{-3}) were approximately 11% higher than at L1 during cross-road winds. Differences were negligible during other wind conditions. The authors attribute this to the impeding effect of vegetation. During cross-road winds, PNCs decreased by 14% and 37% at L3 and L4 respectively, compared to L2. Levels were considerably lower in cross-footpath wind conditions ($1.49 \pm 0.91 \times 10^4 cm^{-3}$ at L4; $1.80 \pm 1.01 \times 10^4 cm^{-3}$ at L3; $6.26 \pm 3.31 \times 10^4 cm^{-3}$ at L2).

Brantley (2014[8], -) examined the effect of trees on air quality around a six lane highway running past a golf course in Detroit in a controlled study. The stand of trees, primarily oak and maple, ranged in width from around 5-78m and were around 10m high, with an underbrush creating a barrier from ground level to tree top. Air quality sampling was carried out using two portable samplers for 28 days in May and June. One sampler was located for the whole period in a clearing around 30m from the highway without any obstruction between it and the road. The second sampler was located around 340m north, at the same distance from the highway and behind a 15 m thick tree stand (measured leaf area index of 3.9). For 2 hours on 13 of the sampling days this sampler was moved every 10 mins sequentially to a series of sampling points located behind the stand of trees and in the clearing. Black carbon and PNC were recorded for different wind conditions.

Results showed black carbon was lower behind the vegetation barrier during downwind and parallel wind conditions. Measurements were 12.4% lower (1.7µgm-3 to 1.49µgm-3) in downwind conditions and 7.8% lower (0.93µgm-3 to 0.85µgm-3) in parallel wind conditions. Reductions were greatest (up to 22% reduction, range not reported) in the late afternoon with winds from the road. Fine ($PM_{2.5}$) and course particle (PM_{10}) counts did not show significant changes during either downwind or parallel wind conditions. (Fine particles changed from $155/cm^3$ to 151 $counts/cm^3$; course particles were unchanged at 8.9 $counts/cm^3$ during downwind conditions. In parallel wind conditions fine particles were unchanged at $76/cm^3$; course particles were $5.5/cm^3$ and $5.7/cm^3$; all not significant). During upwind conditions fine particle numbers were slightly elevated (from 86 to $89/cm^3$ for fine particles,5.7 to $6.1/cm^3$ (not significant). Pollutant concentrations were not modified during low wind conditions.

$PN_{2.0-10.0}$ counts were 8.2% higher at the measuring site behind the vegetation barrier during upwind conditions (5.7 to $6.1/cm^3$) , suggesting a contribution from sources on the golf course side.

The thickness of the stand of trees varied considerable and the authors note a clear relationship between distance and black carbon during downwind but not parallel wind conditions. They note that black carbon reduced by 36% as the distance from the highway increased from 35 to 90 m. This was less than observed in unobstructed areas where a 54% reduction from 30 to 90m was found. They suggest this may be due to slowing of dispersion as pollutants get caught in boundary areas along the edges of the barrier. The authors note that the results may be influenced by the shape of the stand of vegetation (which was roughly triangular) and may not reflect the impact of a continuous stand with similar depths. The authors also note that these results were obtained over the summer when leaf area would be greatest.

Different effects might be expected at other times of the year. They note that when considering the impact of vegetation on air quality from a particular source, wind speed and direction need to be taken into account.

Hagler (2012[18], -) examined the effect of a solid noise barrier and two tree stands (one coniferous, one deciduous) on UFPs near major roads in North Carolina (USA) in a controlled study.

The solid barrier (at Raleigh) was 6m high, 0.5m thick and placed at 5m from the road. Traffic on the road (I-440) was around 108,000 vehicles per day (annual average daily traffic). The coniferous barrier (Chapel Hill) was 6.1±2.3m high, 3.6±1.6m thick at 3.2±0.7m from the road (U.S.Route15–501). Leaf area index varied from 3.3±1 in early autumn to 2.8±1.6 in winter. Traffic volume was 38,000 vehicles per day. The second vegetative barrier was deciduous, at Mebane. It was 7.2±1.3m high, 4.5±1m wide and was placed at 7.7±1.7m from the road. LAI was 3±0.8 in autumn, falling to 1.0±0.5 in winter. The road (I-40/I-85 combined) carries around 85,000 vehicles per day.

Measurements were taken at heights of 3 and 7m behind the barriers and at 3m in clearings with no barrier between the road and measuring site. Measurements were taken at 10m from the road behind the solid barrier (Raleigh), 15m behind the coniferous barrier (Chapel Hill) and 20m behind the deciduous barrier (Mebane).

After accounting for background concentration, the solid (Raleigh) barrier reduced UFP concentrations at 10m from the road by 49-53% (downwind conditions), by 30-61% in parallel wind conditions and by 33-50% with variable winds.

The tree barriers for both sites during autumn had a LAI of around 3. In Chapel Hill, winds during all 3-hour measuring sessions were categorised as parallel, variable or low wind speed so the likely highest impact during downwind conditions was not seen. The UFP concentration at the behind-barrier location was lower than the clearing location during 6 periods, no difference during 5 periods and higher during 5 periods. At Mebane 2 of 3 down-wind measurements of UFPs were lower behind the barrier while the third was higher (this period had relatively high wind direction variability - ($\sigma\theta$=46°, near the defined threshold of 50). 5 out of 9 measurements during other wind conditions showed lower levels of UFPs behind the barrier with 4 showing raised levels behind the barrier. During winter, measurements were generally similar with 6 out of 8 periods having nearly identical concentrations of UFPs at both sites.

The authors note that gaps in the vegetation from irregular tree spacing and branch coverage may allow pollution to pass through. Vegetation may also reduce flow, leading to increased concentrations within and behind the trees under some wind conditions.

Ning (2010[24], -) examined the impact of noise barriers on pollution levels downwind from two highly trafficked freeways in California in a controlled study. A solid barrier may influence air quality by altering the flow of air and so the dispersion of pollutants around the point of release. Two sampling sites were selected on each freeway, one with a roadside noise barrier and one without. Each sampling site used a stationary site at the edge of the freeway to characterise the freeway emissions and a mobile platform to sample ambient air at sites on the downwind trajectory at varying distances. All sites were chosen so that there were no major roadways or industrial sources upwind other than the road under consideration. The noise barrier at one site (I-710) was 3.7m high. The length of the barrier is unclear but appears to be in excess of a hundred metres. The barrier at the second site (I-5) was 5.2m high. Again, the length is unclear but appears to be in excess of a hundred metres. Traffic flow on the I-710 was 12,200 vehicles per hour and 8,500 vehicles per hour on the I-5. Truck flow (heavy duty diesel vehicles) was lower on the I-710 (500 trucks per hour) than the I-5 (640 trucks per hour). (For comparison, the average daily flow on Highways England managed roads is around 81,000 vehicles per day (3,375 per hour) for motorways and 31,000 vehicles per day (1,300 per hour) for A roads).

Without barriers, PNC, especially for smaller (<30nm) particles, decrease exponentially with distance from the freeway, reaching background level (around $3x10^4$particles cm^{-3}) within 120m (for I-5) and 200m (for I-710). The distribution with a barrier is different. At 15m from the freeway particle number concentrations were $4.8x10^4$ and $3.1x10^4$ at I-710 and I-5 respectively, 43% and 45% lower than those measured at 20m downwind of the freeway without a barrier. As the distance from the freeway increases, particle concentrations increase reaching a maximum of around $9x10^4$ particles cm^{-3} for I-5 at 80m and around $1.2x10^5$ particles cm^{-3} for I-710. These figures are around 2.4 and 2.2 times the concentration at the corresponding non-barrier site. Particle concentrations drop from this peak with increasing distance, reaching background levels at around 400m for PNCs. Particle mass concentrations stabilise at background levels earlier, at around 200m. Particles and gaseous co-pollutant concentrations (such as NO_2) increase with increasing distance from the freeway, peaking at 80-100m where the plume of elevated traffic emissions reaches the ground. Levels of particle and co-pollutants concentrations reach background levels at distances of 250-450 m, compared to 150m without barriers, indicating a larger impact zone of traffic emission sources near the freeways with roadside noise barriers.

Baldauf (2016, −) examines the impact of noise barriers on both on-road and downwind pollutant concentrations around a large highway in Phoenix, Arizona, USA.

The noise barriers were approximately 4.5m in height, less than 1m thick, approximately 3m from the nearest travel lane, and had an access road immediately behind the wall.

Measurements of air pollutants (NO_2, UFPs and BC were measured using a mobile platform and fixed sites along two limited access stretches of highway that contained a section of noise barrier and a section with no noise barrier. To maximise the measurement of pollution during 'downwind' conditions measurements were made on one side of the road daily (east or west) based on the forecast wind direction.

Measurements were made within segments on the west side and east side of the highway within 1 km each other. Each segment was approximately 2km in length and 500m in width and primarily residential.

Results indicate that the barriers reduced pollution concentrations for all pollutants measured compared to measurements in areas without barriers. These were largest at distances between 0-50 and 50-150m from the road, although reductions were seen as far as 300m from the road.

Table 14. Median and mean reduction in near-road pollutant concentrations measured under all meteorological and temporal conditions

Pollutant	Sampling section	Distance range (m)	Median reduction (%)	Mean reduction (%)
NO_2	East	0-50	37	37
		50-150	41	39
		150-300	33	28
	West	0-50	34	34
		50-150	20	17
		150-300	19	11
BC	East	0-50	53	43
		50-150	63	49
		150-300	26	18
	West	0-50	57	48
		50-150	55	30
		150-300	37	24
UFP	East	0-50	48	50
		50-150	34	44
		150-300	16	15
	West	0-50	54	66
		50-150	27	31
		150-300	12	23

Evidence statement 4.1: natural and artificial barriers influence distribution of air pollutants from major roads

Weak evidence from 5 studies (4 US[8,24,18,30] and 1 from the UK[3] (all -) examined natural or artificial barriers. Overall the evidence suggested that barriers are effective in changing the distribution of particulate pollutants from major roads, although the details will vary with specific factors. Factors include shape and permeability of the barrier and local meteorological conditions.

4.1a Three studies[24, 18, 30] showed a substantial near-barrier reduction in particle numbers concentrations from solid barriers with a potentially negative effect at a greater distance (80m-400m). One study[24] found a 43-45% reduction at 20m from the barrier and a 2.2-2.4 fold increase at 80m, dropping to background levels at around 400m. The second study[18] found concentrations of ultra-fine particles (UFP) were reduced at 10m from the barrier by 49-53% for downwind conditions, 30-61% for parallel to road wind and 33-50% for variable winds. The mean reduction for non-upwind conditions was 47%. The third study[30] found reductions in number of UFPs of between 50-66% at 0-50m and a reduction of between 15-23% at 150-300m. Similar results were found for black carbon (43-48% reduction at 0-50m; 18-24% at 150-300m) and NO_2 (34-37% at 0-50m; 11-28% at 150-300m).

4.1b Three studies[8, 3, 18] looked at natural barriers. Two US[8, 24] (-) studies suggest a small or negligible effect of stands of trees on particles under most wind conditions; a single UK (-) study suggested substantial (up to 37%) reductions in UFPs near a major road from dense line of bushes under some wind conditions.

Brantley (2014)[8] examined the effect of a stand of deciduous trees around 5-78m wide and around 10m high on pollution from a 6-lane highway. Results showed significant reductions of 12.4% (from 1.7μgm-3 to 1.49μgm-3) and 7.8% (from 0.93μgm-3 to 0.85μgm-3) in black carbon during downwind and parallel wind conditions. The reduction due to distance from the source (36% decrease from 35 to 90 m) was less than the reduction noted by others in unobstructed areas (a 54% reduction from 30 to 90m), suggesting a slowing of dispersion as pollutants get caught in boundary areas along the edges of the barrier. Fine ($PM_{2.5}$) and course (PM_{10}) particle counts did not show significant changes during either downwind or parallel wind conditions.

Hagler (2012)[18] found no consistent reduction in concentrations of UFPs behind two stands of trees (approximately 6-7m high by 3.6-4.5m thick), compared with levels measured in a clearing.

Al Dabbous (2014)[3] looked at the effect of a 2.2m wide dense hedge next to a major road on particle numbers on an adjacent footpath. Effects varied with wind direction, with the largest effect seen during cross-road winds. PNCs decreased gradually through the vegetation barrier, reaching a maximum reduction of 37% on the footpath compared to levels immediately next to the road during cross-wind conditions.

Applicability: Both studies were carried out in the US and examined situations which could be replicated in England so the evidence is applicable.

8. Brantley et al. 2014 (−)

24. Ning et al. 2010 (−)

3. Al-Dabbous et al. 2014 (−)

18. Hagler et al. 2012 (−)

30. Baldauf et al. 2016 (−)

4.3.2 Use of dust suppressants

Amato (2014[4], -) looked at the daily application of CMA (calcium magnesium acetate) or $MgCl_2$ (hygroscopic materials to which particulate matter adheres, preventing its resuspension) in a controlled before and after study. The full width (9m) of a road in the commercial district of Barcelona (4000 vehicles/day) was treated daily with CMA (25% aqueous solution) or $MgCl_2$ (20% aqueous solution). Applications were made between 5 and 9am and the road cleaned using a vacuum-assisted wet sweeping vehicle after 23:00 to prevent build-up of materials. Because of the possible influence of CMA on friction and hence road safety measures were taken to reduce the speed limit from 50kph to 30 kph. Warnings about slippery conditions were also given. Three application phases were used (phase I – CMA over a 1400m stretch for 3 days (April 16-18); phase II – CMA over 2300m on 3 days (April 22-24) and subsequently 4 days (May 2, 6, 8, 13); phase III – $MgCl_2$ over 2300 for 2 days (May 21, 23)). Days were chosen that were forecast to be dry with temperatures above 0°C. Air quality measurements were made at five sites: three mobile vans in the parking lane of the treated road, one control kerbside van in a parallel untreated road and one background control site 5.5km from the treated road.

During phase I only one measuring site could be used due to construction work nearby. Phase I also coincided with a Saharan dust episode which raised PM_{10} levels in NE Spain by 3-9 μgm^{-3}. The authors note that the high deposition from this source may have severely affected the efficacy of CMA and so phase I may not be representative. During phase II, PM_{10} and $PM_{2.5}$ decreased at only one of the sites (-1 μgm^{-3} difference) but only in relation to the urban background site. This reduction was not statistically significant ($p>0.05$). During phase III, a small, non-significant reduction of 3-4 μgm^{-3} in PM_{10} was seen at one measuring site and a non-significant decrease in $PM_{2.5}$ at 1 site in relation to the urban background site.

Gillies (1999[14], -) looked at the efficiency of four dust suppressants on reducing the emission of PM_{10} from unpaved roads in California in a controlled study. The suppressants were: a biocatalyst stabiliser (BS) which is intended to increase the cementation and stability of compacted aggregate and earth materials; a polymer emulsion (PE) designed to create a surface film that seals the underlying material; a petroleum emulsion with polymer (PEP) which bonds the road material and forms a surface crust; and a non-hazardous crude oil containing material (NHCO) which forms a hard, pliable surface of cemented aggregates. Each treatment was applied to a 500m stretch of a straight road 3km in length. A fifth section was left untreated for comparison. PM_{10} measurements were taken at heights from 1.25-9m above ground level and at 30m from the roadway on both sides of the road. Measurements were also taken at 9m above each section of the road. PM_{10} emissions were created using a ¾ ton pick-up truck travelling along the roadway for 100 passes. The truck speed was kept constant at either 40 or 55km/hr. Vehicle kilometres travelled by local traffic was recorded by traffic counters and included in the total vehicle kilometres travelled by the pick-up. One week after application, suppressant efficiencies ranged between 33% and 100% for the four types applied. After weathering, efficiencies had dropped to between zero and 92%.

Suppressant	Efficiency at 1 week	Efficiency after weathering
BS	33% (±26%)	Zero
PE	94% (±6%)	86% (±5%) [after 11 months]
PEP	99% (±2%)	49% (±10%) [after 11 months]

Evidence statement 4.2 : dust suppressants

Weak evidence from 2 [-] studies[2,7] suggested no significant reductions in particulates from the use of dust suppressants (CMA and $MgCl_2$) on urban roads, with a larger, short term effect from sealant treatments on unpaved roads.

Amato (2014)[4] looked at the application of CMA and $MgCl_2$ on a road in Barcelona carrying 4000 vehicles/day. No significant reductions in PM_{10} or $PM_{2.5\text{-}10}$ were seen. A small, non-significant reduction in PM_{10} (1µg/m^3) was seen at 1 of 3 sites compared with one of 2 control sites with application of CMA and at 1 of 3 sites with $MgCl_2$ (3µG/m^3). Non-significant reductions in $PM_{2.5\text{-}10}$ (2 to 3 µg/m^3) were seen with both CMA and MgCl2 applications compared with one control site.

Gillies (1999)[14] looked at the efficiency of a biocatalyst stabiliser (BS); a polymer emulsion (PE); a petroleum emulsion with polymer (PEP); and a non-hazardous crude oil containing material (NHCO) on an unpaved road in California. After 11 months of exposure to weathering and 4,900–6,400 vehicle passes, the suppressant efficiencies for PM_{10} were 49% (±10%) for PEP and 86% (±5%) for PE. After eight months' aging, the NHCO suppressant's efficiency was 92% (±6%). Efficiency for BS had fallen to zero, producing similar levels to the untreated section.

Applicability: One study (Amato)[2] was from Spain and one from the US (Gilles[7], from California). There are considerable differences in weather conditions between Spain (Barcelona) and England and between the US (California) and England which would have a substantial impact on the outcomes. The study by Gilles looked at an unpaved road with little traffic, a situation which is uncommon in England. Therefore the evidence has limited applicability.

4. Amato 2014 (−)

14. Gilles 1999 (−)

4.3.3 Use of street washing and sweeping

Amato (2009[5], -) looked at the effectiveness in reducing particulate matter of street washing in Barcelona in a controlled before and after study. The street in question was a busy mixed commercial and residential street with a mean traffic flow of 19,000 vehicles per day on a five-lane one-way highway. Street washing used 0.95l/m^2 water, preceded for the last three washings by a mechanical road sweeper vehicle. Eight washing cycles were carried out. PM_{10}, together with meteorological data were recorded at two sites 1200m apart (one upwind and one downwind) and at 4 background sites belonging to the local air quality monitoring network. On street washing days average daily concentration of PM_{10} decreased by 8.8μgm^{-3} at the downwind site compared with a reduction of 3.7μgm^{-3} at the upwind site. The morning peak was similarly reduced at the downwind site compared to the upwind site on street washing days (81 and 96μgm^{-3} respectively) compared with non-washing days (73 and 76 μgm^{-3} respectively). The authors suggest this indicates either a decrease from washing at both sites, enhanced at the downwind site due to wind transference or a regional reduction due to meteorological factors, enhanced at the downwind site due to street washing. They also note that data from background monitoring sites suggest a reduction of 3.7-4.9 μgm^{-3} due to meteorological factors in the study area and a further 4-5 μgm^{-3} (7-10%) of kerbside PM_{10} in the 24 hours after treatment.

Evidence statement 4.3: Effectiveness of street washing and sweeping

Weak evidence from 1 [-] study (Amato (2009[5])) suggests a small positive effect in reducing particulate matter from street washing.

The street in question (in Barcelona) was a busy mixed commercial and residential street with a mean traffic flow of 19,000 vehicles per day on a five-lane one-way highway. On street washing days average daily concentration of PM_{10} decreased by 8.8μgm^{-3} at the downwind site compared with a reduction of 3.7μgm^{-3} at the upwind site.

Applicability: the study was carried out in Barcelona during April and May on days forecast to be dry. The authors note the substantial impact of rain on deposited dust, therefore the UK climate may limit the transferability of the evidence.

5. Amato 2009 (−)

4.3.4 Urban greening

Green infrastructure (such as trees, hedges, rooves and walls covered with plants of various types) can influence air pollution in a number of ways. These include altering the flow of air through streets (and so the dilution of pollutants) and by slowing pollutants and making it more likely that they will be deposited on the leaves.

Pugh (2012[25], -) modelled the impact of green infrastructure on the air quality of urban street canyons. The authors used a development of the atmospheric chemistry model CiTTyCAT called CiTTy-Street which can assign deposition velocities for roofs, canyon walls, and floor separately. This model was used to calculate the effects of urban vegetation on pollutant concentrations using Central London as a case study. It considers one control and three green wall/green roof scenarios, each for a single street canyon and for a large area of street canyons. Green walls in the context of this paper are any in-canyon vegetation which minimally affects in-canyon residence time. The effect of vegetation on deposition velocities was set in the middle of the most commonly reported values from the literature (0.3cm s-1) rather than selecting a deposition velocity for a specific species.

Adoption of green walls on large areas of street canyons resulted in a reduction of NO_2 and PM_{10} of up to 15% and 23% respectively (wind speed 1 ms-1, canyon height to width (h/w) ratio 1). The reduction was dependent on residence time (dependent on wind speed and canyon geometry) and the fraction of canyon wall greened but not the initial pollutant concentration. The net pollutant flux out of the canyon was reduced by 2-11% for NO_2 and became inward for PM_{10}, leading to small concentration reductions in the urban boundary layer. The authors note that as cities are major regional sources of air pollutants this effect could be significant for pollutant transport and regional scale photochemistry for cities with large areas of street canyons such as London or Paris.

Table 15. Modelled vegetation scenarios and expected in-canyon concentration reductions under different canyon configurations and meteorological conditions

			Concentration change relative to control scenario (%)			
			Wind speed = 2ms-1			**Wind speed = 0.5ms-1**
	Deposition velocities (cm s-1)		**Aspect ratio = 1 (h/w ratio)**		**Aspect ratio = 2 (h/w ratio)**	
	NO_2	**PM_{10}**	**Numerous canyons**	**Single canyon**		
Green walls (100%)	Walls: 0.3	Walls: 0.64	NO_2: -8.9% PM_{10}: -13.1%	NO_2: -6.4% PM_{10}: - 10.8	NO_2: -19.9% PM_{10}: -32.0%	NO_2: -42.9% PM_{10}: -61.9%
	Roof: 0.05	Roof: 0.2				
Green roof	Walls: 0.05	Roof: 0.02	NO_2: -0.9 PM_{10}: -1.1			
	Roof: 0.3	Roof: 0.64				

The authors note that for surfaces with comparable leaf indexes (and hence deposition velocities) greening in-canyon surfaces is more effective at reducing street-level pollutant concentrations than green roofs as it acts on the relatively small volume of air inside the canyon rather than via the urban boundary layer.

To evaluate the effect over realistic wind speeds across a year the authors used data for the daily average wind speeds at Kew (London) across 2008. Using an idealised city of uniform street canyons with a height to width ratio of 1, annual average concentrations of NO_2 and PM_{10} were reduced by 9% and 13% respectively by greening of canyon walls across large areas. Reductions for a single canyon were

7% and 11%, increasing to 20% and 31% when the height/width ratio was increased to 2.

The authors also examined the effect of street trees. Like green walls, trees can increase deposition but they also reduce mixing between street canyon air and the urban boundary layer. To examine this, the authors carried out sensitivity analyses by altering deposition velocity and canyon residence time. They note that in highly polluted canyons trees may reduce levels of PM_{10} but increase concentrations of NO_2 in most circumstances.

Vos (2013[28], +) modelled the effect of trees, hedges and green barriers on a variety of traffic related pollutants. It focuses on a variety of non-street canyon cases and real life geometries.

Simulations were carried out using the ENVI-met model, a three dimensional computational fluid dynamics model tailored for simulating different urban atmospheric process such as dispersion and microclimate effects. Sensitivity analyses are used to investigate how different parameters influence the effect of vegetation on local air quality. Four parameters were varied: the building geometry, the type of pollutant, the meteorological conditions and the vegetation. The building geometries used were selected to represent the types found in Belgium and the Netherlands (a street canyon and a detached housing geometry). Wind direction was varied (wind speed was fixed at $3ms^{-1}$) so that it was either perpendicular to the street or oblique. Parameters for the vegetation include type (trees, hedges, green barriers), height, location, porosity and filtering capacity. In all, 17 vegetation scenarios were used. A total of 144 different scenarios were examined. Air quality at the footpath was used as the measure of interest.

Summary data is not presented in the paper, which relies on graphical displays from which data could not be extracted. Overall, the authors note that trees have less influence on PM_{10} than NO_2, mainly due to the higher contribution to PM_{10} levels of non-traffic sources. The general effect of vegetation was similar for both geometries examined, except that there was little effect seen on the leeward side for detached houses. Qualitatively the impact was similar for both wind directions.

The authors note that trees significantly increase pollutant concentrations, and a deterioration in air quality is also seen with hedges. Green barriers, which represent an impermeable core, appeared to improve air quality at the footpath. This effect is also seen with bare solid screens without vegetation cover.

Increasing the height of hedges or barriers alters the findings in different directions. A higher barrier leads to lower concentrations at the footpath (due to better shielding) while a higher hedge worsens air quality (due to reduced ventilation).

Fewer trees (greater distance between trees) results in lower pollutant concentrations.

Simulations with five times higher deposition speeds than the default show no significant difference in results, suggesting that it is mainly the aerodynamic effect that determines the overall impact on air quality rather than the pollutant removal capacity.

The authors also examined 19 case studies which represent real-life designs used to try to improve air quality using vegetation. All 19 are based on streets in Belgium and the Netherlands. For each case the effect was modelled with and without the proposed vegetation, using a perpendicular wind direction. Only one design (a 4m high impermeable green barrier between traffic and the footpath) improved air quality. The other designs, involving trees, had detrimental effects on air quality. This was greater with higher surrounding buildings, situations which were likely to have higher air pollution levels even without trees.

The authors note that the scope of the work was to examine the impact of roadside urban vegetation on local air quality. The overall impact of vegetation on city-averaged air quality is likely to be positive.

Vranckx (2015[29], -) modelled the effect of urban trees in a street canyon using the OpenFOAM CFD package. They examined two different street canyon widths with width/height rations of 1.0 and 2.0. Wind directions from 0° to 90° in 15° steps were examined. For each wind direction, vegetation parameters were varied to represent different leaf areas and porosities. A total of 70 simulations were run to produce average normalised concentrations near the buildings in the idealized street canyon.

These simulations were used with the meteorological data from 2012 from a station near Antwerp to find the annual average concentration of PM_{10}. The effect of wind direction was considered in four orientations (along the cardinal and intercardinal directions). To allow for changes in the leafiness of trees across the year, results for streets with no trees were used to simulate conditions between 1st Nov and 15th April, and results with trees for the summer period.

Hourly background concentrations for PM_{10} for Antwerp were used to allow consideration of background pollution as well as sources within the canyon. For PM_{10}, emissions from within the canyon contribute approximately 7.5% of the total. The effect of trees on PM_{10} is an increase of 0.2% to 2.6%, depending on orientation and type of vegetation.

The authors note that even where local emissions inside a canyon make up 40% of the pollutant concentrations (for instance for EC in Antwerp), introduction of trees would lead to increases in annual concentrations of up to 8%.

Evidence statement 4.4 Adverse effect of street trees

Weak evidence from three [-] quality modelling studies (2 from Belgium[28], [29], one from the UK[25]) demonstrate the risk of an adverse effect from street trees in urban canyons in most circumstances. Vegetation that does not reduce air flow may improve air quality.

Adoption of green walls which do not reduce air flow on large areas of street canyons resulted in a reduction of NO_2 and PM_{10} from 0 % (no green wall coverage) up to 15% and 23% respectively[25] at 100% wall coverage with vegetation. In 19 simulations based on streets in Belgium and the Netherlands only one design (a 4m high impermeable green barrier between traffic and the footpath) improved air quality. All other designs had detrimental effects on air quality within the streets[28].

The annual effect of trees on PM_{10} in an urban canyon ranges from increase of around 0.2% to an increase of 2.6%, depending on orientation and type of vegetation [29].

Applicability: The studies are from Belgium and the UK and so are partially applicable.

25. Pugh, 2012 (−)

28. Vos, 2013 (−)

29. Vranckx, 2015 (−)

5 Discussion

5.1 Strengths and limitations of the review

Overall, the quality of the evidence reviewed was poor. Twenty three of 29 studies scored [-]. The majority of studies did not include health effects as outcomes (Jarjour[20] and Burr[10] included impacts on lung function; Chong[11] modelled impact on mortality). However this is in keeping with most studies on the impact of interventions on air pollution.

Three cost effectiveness studies were included, all rated as moderate [+] quality. Two of these (using the same methodology) examined changes in the fuelling of bus fleets in the US. One looked at the use of diesel hybrid vehicles for deliveries.

Six studies of cycle routes were only able to measure levels of pollutants on a small number of routes which may be influenced by local factors such as street design and topography. However, the results tended to show changes in a consistent direction. In other areas the review found only small (one or two) studies which examine complex areas such as changes to public transport fleets. One such study (Gramsch[17]) used measurements on a small number of streets where there were several different changes in transport which may have influenced the results. A second public transport study (Burgard[17]) looked at two fleets of buses where performance may have been influenced by road conditions. They were unable to examine changes before and after installation of new technologies. Four modelling studies looked at the impacts of changing fuels in public transport fleets. While these provide useful indications of the potential impacts, modelling studies are inherently at risk of bias due to the nature and extent of assumptions and simplifications involved. A further 3 studies considered the economic impacts of changes to the fuels used in vehicle fleets. Again, there are simplifications and assumptions in these. The study of a bypass construction was potentially influenced by selection bias from people moving away, possibly due to health issues. Four studies examined barriers near roads. The influence of particular local factors (such as season, density of planting, height of barriers, local topography and meteorological conditions and source of emissions) means that care needs to be taken in extrapolating to other situations.

Two studies looked at the impact of dust suppressants used in urban roads. These were carried out in Barcelona (Amato[4]) and California (Gillies[14]). One further study looked at street washing in Barcelona (Amato[5]). They were also carried out on a limited number of sites and so are at risk from the influence of local factors, including weather and the use of unsealed roads.

No studies were found which examined the measured effect of street trees on air quality in urban environments. As a result, 3 modelling studies were included (Pugh[25], Vos[28] and Vranckx[29]). While these provide useful indications of likely impact as they are modelling studies they should be treated with caution as there is no verification of the findings.

Studies that were based on measurements of air pollutants tended not to report the extent of variation of the measurement technique used, and an examination of this is beyond the scope of this work. However it should be noted that this may be considerable and may be larger than some of the results reported.

Measurements tended to be made in a relatively limited number of locations which may be influenced unduly by local conditions (for instance wind direction).

It should be noted that these generic considerations about strengths and weaknesses apply to other evidence gathered for this guideline.

Further detail of the strengths and weaknesses of individual studies can be found in the evidence tables (Appendix 4).

5.2 Applicability

Studies were from Chile (1), Spain (4), Canada (3), Greece (1), UK (5), Netherlands (1), Belgium (2) and the US (11). Local factors will be particularly significant in some cases thus reducing the applicability of studies. However, it should also be noted that changes in vehicle fleet composition over time will also have had an effect. Thus, the studies are in general partially applicable.

5.3 Gaps in the evidence

5.3.1 Review question 2: Are interventions to develop public transport routes and services, effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

We set out to find evidence of the effectiveness of:

- Implementation or changes to bus or public transport lanes
- Implementation of or changes to public transport services (including cost)
- Public transport quality improvements
- Use of standards in commissioning public transport services
- Provision of information about existing services
- Action to integrate public transport services with other low emission modes such as walking or cycling

No UK evidence was found relating to these questions was found.

5.3.2 Review question 3: Are interventions to develop routes and infrastructure to support low emission modes of transport effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

We set out to find evidence of the effectiveness of:

- Implementation of or changes to cycle routes or pedestrianised areas
- Implementation of or changes to fuelling services for low emission vehicles
- Use of low emission public sector vehicle fleets
- options for siting of routes (e.g. low traffic vs normal traffic; avoiding inclines; siting and timing of traffic signals)

No evidence was found relating to low emission vehicles and limited evidence was found for low emission public sector vehicle fleets.

5.3.3 Review question 4: Are measures to promote absorption, adsorption or impingement deposition, and catalytic action effective and cost effective at reducing the health impact of, or people's exposure to, traffic-related air pollution?

We set out to find evidence of the effectiveness of measures to promote absorption, adsorption or impingement, deposition and catalytic action on reducing the health impact of and people's exposure to traffic-related air pollution.

While a limited number of studies were found that looked at measures that might affect impingement and deposition (natural and artificial barriers) these were limited in their extent and nature.

- No studies were found which looked at the impact of street trees and greenery,
- No studies were identified measuring the impact in specific locations such as traffic 'canyons'.

As a result, modelling studies relating to street trees and greenery were included.

Studies were found which looked at the impact of a range of surface treatments aimed at reducing levels of particulates and particulate resuspension however:

- No studies were found which looked at catalytic action, such as titanium dioxide.
- No studies were found specifically from the UK on the effect of surface treatments, or from countries with similar meteorological conditions.

6 Included studies

1. Alam, Ahsan, Diab, Ehab, El-Geneidy, Ahmed M., Hatzopoulou, Marianne, A Simulation of Transit Bus Emissions along an Urban Corridor: Evaluating Changes under Various Service Improvement Strategies, Transportation Research: Part D: Transport and Environment, 31, 189-98, 2014

2. Alam, Ahsan, Hatzopoulou, Marianne, Reducing transit bus emissions: Alternative fuels or traffic operations?, Atmospheric Environment, 89, 129-139, 2014

3. Al-Dabbous, A. N., Kumar, P., The influence of roadside vegetation barriers on airborne nanoparticles and pedestrians exposure under varying wind conditions, Atmospheric Environment, 90, 113-124, 2014

4. Amato, Fulvio, Karanasiou, Angeliki, Cordoba, Patricia, Alastuey, Andres, Moreno, Teresa, Lucarelli, Franco, Nava, Silvia, Calzolai, Giulia, Querol, Xavier, Effects of road dust suppressants on PM levels in a Mediterranean urban area, Environmental Science & Technology, 48, 8069-77, 2014

5. Amato, F., Querol, X., Alastuey, A., Pandolfi, M., Moreno, T., Gracia, J., Rodriguez, P., Evaluating urban PM_{10} pollution benefit induced by street cleaning activities, Atmospheric Environment, 43, 4472-4480, 2009

6. Bean, T., Carslaw, N., Ashmore, M., Gillah, A., Parkinson, C., How does exposure to nitrogen dioxide compare between on-road and off-road cycle routes?, Journal of Environmental Monitoring, 13, 1039-1045, 2011

7. Boogaard, H., Borgman, F., Kamminga, J., Hoek, G., Exposure to ultrafine and fine particles and noise during cycling and driving in 11 Dutch cities, Atmospheric Environment, 43, 4234-4242, 2009

8. Brantley, H. L., Hagler, G. S. W., J. Deshmukh P, Baldauf, R. W., Field assessment of the effects of roadside vegetation on near-road black carbon and particulate matter, Science of the total environment, 468-469, 120-129, 2013

9. Burgard, D. A., Provinsal, M. N., On-road, in-use gaseous emission measurements by remote sensing of school buses equipped with diesel oxidation catalysts and diesel particulate filters, Journal of the Air and Waste Management Association, 59, 1468-1473, 2009

10. Burr, M. L., Karani, G., Davies, B., Holmes, B. A., Williams, K. L., Effects on respiratory health of a reduction in air pollution from vehicle exhaust emissions, Occupational and environmental medicine, 61, 212-218, 2004

11. Chong, U., Yim, S. H. L., Barrett, S. R. H., Boies, A. M., Air quality and climate impacts of alternative bus technologies in Greater London, Environmental Science and Technology, 48, 4613-4622, 2014

12. Cohen, Joshua T., Diesel vs. compressed natural gas for school buses: a cost-effectiveness evaluation of alternative fuels, Energy Policy, 33, 1709-1722, 2005

13. Cohen, J. T., Hammitt, J. K., Levy, J. I., Fuels for urban transit buses: A cost-effectiveness analysis, Environmental Science & Technology, 37, 1477-84

14. Gillies, J. A., Watson, J. G., Rogers, C. F., DuBois, D., Chow, J. C., Langston, R., Sweet, J., Long-term efficiencies of dust suppressants to reduce PM_{10} emissions from unpaved roads, Journal of the Air and Waste Management Association, 49, 3-16, 1999

15. Goncalves, M., Jimenez-Guerrero, P., Baldasano, J. M., High resolution modeling of the effects of alternative fuels use on urban air quality: Introduction of natural gas vehicles in Barcelona and Madrid Greater Areas (Spain), Science of the total environment, 407, 776-790, 2009

16. Goncalves, Maria, Jimenez-Guerrero, Pedro, Baldasano, Jose M., Emissions variation in urban areas resulting from the introduction of natural gas vehicles: application to Barcelona and Madrid greater areas (Spain), The Science of the total environment, 407, 3269-81, 2009

17. Gramsch, E., Le Nir, G., Araya, M., Rubio, M. A., Moreno, F., Oyola, P., Influence of large changes in public transportation (Transantiago) on the black carbon pollution near streets, Atmospheric Environment, 65, 153-163, 2013

18. Hagler, G. S. W., Lin, M. Y., Khlystov, A., Baldauf, R. W., Isakov, V., Faircloth, J., Jackson, L. E., Field investigation of roadside vegetative and structural barrier impact on near-road ultrafine particle concentrations under a variety of wind conditions, Science of the total environment, 419, 7-15, 2012

19. Hatzopoulou, Marianne, Weichenthal, Scott, Dugum, Hussam, Pickett, Graeme, Miranda-Moreno, Luis, Kulka, Ryan, Andersen, Ross, Goldberg, Mark, The impact of traffic volume, composition, and road geometry on personal air pollution exposures among cyclists in Montreal, Canada, Journal of Exposure Science & Environmental Epidemiology, 23, 46-51, 2013

20. Jarjour, S., Jerrett, M., Westerdahl, D., De Nazelle, A., Hanning, C., Daly, L., Lipsitt, J., Balmes, J., Cyclist route choice, traffic-related air pollution, and lung function: A scripted exposure study, Environmental Health: A Global Access Science Source, 12, 2013

21. Kendrick Christine, M., Moore, Adam, Haire Ashley, Raye, Bigazzi Alexander, York, Figliozzi, Miguel, Monsere Christopher, M., George, Linda, Transportation Research Board, Fifth Street N. W. Washington D. C. U. S. A., Impact of Bicycle Lane Characteristics on Bicyclists' Exposure to Traffic-Related Particulate Matter, Transportation Research Board 90th Annual MeetingTransportation Research Board, 15

22. Krutilla, Kerry, Graham, John D., Are Green Vehicles Worth the Extra Cost? The Case of Diesel-Electric Hybrid Technology for Urban Delivery Vehicles, Journal of Policy Analysis and Management, 31, 501-32, 2012

23. MacNaughton, P., Melly, S., Vallarino, J., Adamkiewicz, G., Spengler, J. D., Impact of bicycle route type on exposure to traffic-related air pollution, Science of the total environment, 490, 37-43, 2014

24. Ning, Z., Hudda, N., Daher, N., Kam, W., Herner, J., Kozawa, K., Mara, S., Sioutas, C., Impact of roadside noise barriers on particle size distributions and pollutants concentrations near freeways, Atmospheric Environment, 44, 3118-3127, 2010

25. Pugh, Thomas A. M., Robert MacKenzie, A., Duncan Whyatt, J., Nicholas Hewitt, C., Effectiveness of Green Infrastructure for Improvement of Air Quality in Urban Street Canyons, Environmental Science & Technology, 46, 7692-7699, 2012

26. Soret, A., Guevara, M., Baldasano, J. M., The potential impacts of electric vehicles on air quality in the urban areas of Barcelona and Madrid (Spain), Atmospheric Environment, 99, 51-63, 2014

27. Stamos, Iraklis, Kitis, George, Basbas, Socrates, The implementation of a contra flow bus lane in the city of Thessaloniki: Energy and Environmental Impacts, Fresenius Environmental Bulletin, 22, 2191-2196, 2013

28. Vos, Peter E. J., Maiheu, Bino, Vankerkom, Jean, Janssen, Stijn, Improving local air quality in cities: to tree or not to tree?, Environmental pollution (Barking, Essex: 1987), 183, 113-22, 2013

29. Vranckx, S., Vos, P., Maiheu, B., Janssen, S., Impact of trees on pollutant dispersion in street canyons: A numerical study of the annual average effects in Antwerp, Belgium, Science of the total environment, 532, 474-483, 2015

30. Baldauf, R., Isakov, V., Deshmukh, P., Venkatram, A., Yang, B., Zhang, K., Influence of solid noise barriers on hear-road and on-road air quality. Atmospheric environment, 129, 265-276, 2016.