Review of Urban Bicyclists' Intake and Uptake of Traffic-Related Air Pollution

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Abstract

Bicycling as a mode of transportation is enjoying a boost in many urban areas around the world. Although there are clear health benefits of increased physical activity while bicycling, bicyclists may experience increased inhalation of traffic-related air pollutants. Bicyclists have two to five times higher respiration rates than travelers in motorized vehicles and this difference increases with bicycle travel speed and exertion level. The main goal of this work is to review the state of knowledge regarding urban bicyclists' intake and uptake of traffic-related air pollution and to identify key knowledge gaps. This review includes not only bicyclists' exposure to air pollution concentrations but also respiration rates, intake doses (the amount of pollutant that is inhaled), and uptake doses (the amount of pollutant that is incorporated into the body). Research gaps and opportunities for future research are discussed. This is the first review to specifically address bicyclists' health risks from traffic-related air pollution and to explicitly include intake and uptake doses in addition to exposure concentrations for travelers.

Keywords: review, bicycle, pollution exposure, inhalation, uptake, health

1 Introduction

Bicycling as a mode of transportation is enjoying a boost in urban areas around the world through new bike-sharing systems, bicycle-specific roadway facilities, public outreach and incentive programs (Pucher & Buehler, 2012). The push toward promoting bicycling is motivated by a range of environmental, economic, health, and social benefits. Although there are clear health benefits of increased physical activity, bicyclists may experience increased inhalation of traffic-related air pollutants (de Hartog, Boogaard, Nijland, & Hoek, 2010).

Human exposure to traffic-related air pollution has well-established negative health impacts for urban populations (Brook et al., 2010; Forastiere & Agabiti, 2013; Health Effects Institute, 2010; Nawrot et al., 2011). Air pollution exposure is particularly high for travelers because of proximity to mobile sources of pollution (Kaur, Nieuwenhuijsen, & Colvile, 2007), and air quality is a source of concern for urban bicyclists (Badland & Duncan, 2009). However, the health risks of air pollution exposure during travel are not easily characterized because of the numerous individual, environmental, and traffic factors involved.

A conceptual diagram linking traffic-related pollution emissions and health effects is illustrated in Figure 1, adapted from Ott, Steinemann, & Wallace (2007). Motor vehicle emissions (a) degrade urban air quality (b) in accordance with atmospheric dispersive, chemical, and physical processes. Travelers' exposure concentrations (c) then depend on their travel trajectory. The inhalation of traffic-related air pollution (d) depends on travelers' breathing volume while exposed to a pollutant concentration. Uptake of the inhaled pollutants into the body (f) depends on processes in the respiratory tract and other body systems. Finally, the health effects (g) of air pollution uptake doses are a function of the toxicity of the pollutants and physiology of the individual. The processes between inhalation and uptake can be further demarcated as (e_1) intake dose (the amount of pollutant that crosses the body boundary at the mouth and nose), (e_2) absorbed dose (the amount of pollutant that is not exhaled but deposited or absorbed), (e_3) effective dose (the bioavailable amount of pollutant that reaches body tissue instead of being expelled from the respiratory tract lining by coughing, sneezing, etc.), and (e_4) uptake dose (the amount of pollutant that is incorporated into the body).

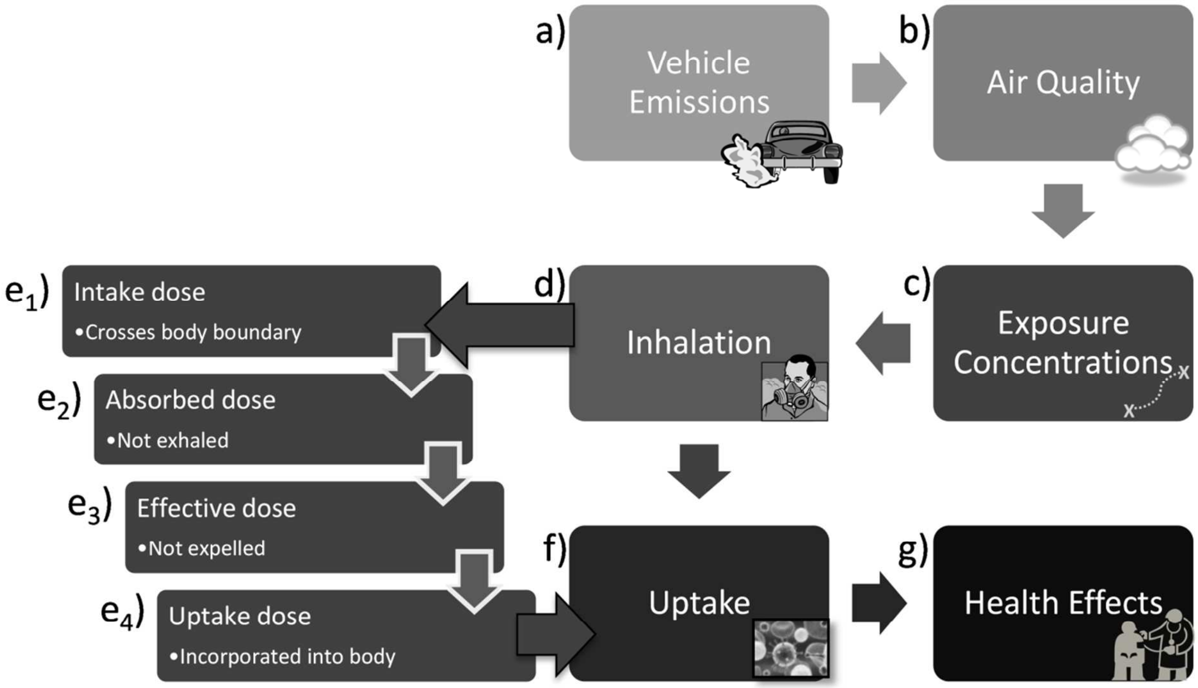


Figure 1. Conceptual Diagram of Exposure Pathway for Traffic-Related Air Pollution

Past reviews of travelers' pollution exposure have been oriented by pollutant (Kaur et al., 2007; Knibbs, Cole-Hunter, & Morawska, 2011) and/or focused on in-vehicle exposures (El-Fadel & Abi-Esber, 2009). These reviews focused on exposure concentrations and provide little or no discussion of respiration or its effects on intake and uptake doses. The focus of this review is on bicyclists' exposure to, inhalation of, and uptake of traffic-related air pollution – i.e. steps (c) through (f) in Figure 1. This review is unique in focusing exclusively on bicyclists.

2 Methodology

A systematic literature search for bicyclist exposure and dose measurements was performed through January 2014 using all 20 possible keyword combinations $\{A + B + C\}$ utilizing the keyword sets $A = \{\text{bicycle, bicyclist, cyclist, bike}\}$, $B = \{\text{pollution}\}$ and $C = \{\text{exposure, intake, inhalation, uptake, dose}\}$. An exhaustive search was performed using the WorldCat™ catalogue. Additional references were found by reviewing cited reference lists and the Google Scholar™ search engine. There were 57 published papers describing original studies of on-road bicyclists and air pollution exposures with spatially-explicit concentration data. There were 42 published papers with unique exposure concentration data measured on-road by bicyclists. Details of the literature search method are presented in the Supplemental Material.

3 Bicyclists' Air Pollution Exposure Concentrations

The main traffic-related air pollutants linked to health risks for road travelers and measured for bicyclists are carbon monoxide (CO), nitrogen oxides (NO_x) – including nitric oxide (NO) and nitrogen dioxide (NO_2), volatile organic compounds (VOC), and particulate matter (PM) of various sizes and composition: ultrafine particles (UFP), $PM_{2.5}$, PM_{10}, and elemental carbon (EC) / black carbon (BC). These pollutants are described in the Supplemental Material.

A traveler's exposure concentration is the concentration of pollutants in their breathing zone. Concentrations of traffic-related primary pollutants are particularly high near roadways – especially for shorter-lived pollutants such as UFP and reactive VOC (Gordon et al., 2012; Karner, Eisinger, & Niemeier, 2010). Steep concentration gradients can be seen even on the scale of a few meters (Clifford, Clarke, & Riffat, 1997; McNabola, Broderick, & Gill, 2009b; Tiwary, Robins, Namdeo, & Bell, 2011). Exposure concentrations are sampled using a variety of pollutant-specific devices, each requiring specialized knowledge and careful sampling procedures (Vallero, 2007). Roadside studies of air pollution concentrations are more common than on-road data collections because on-road measurements are more difficult to execute (particularly for pedestrians and bicyclists). But the body of research on active travelers' pollution exposure concentrations has grown notably in recent years. On-road air quality sampling has become more precise and more portable because of improvements in measurement technology, power storage, and position tracking systems (Gulliver & Briggs, 2004; Steinle, Reis, & Sabel, 2013).

A literature search revealed 42 published studies reporting unique exposure concentration data collected with on-bicycle sampling devices. Summary information on all 42 studies is included in the Supplemental Material, allowing comparisons of methodologies and settings. Table 1 summarizes reported concentrations in all 42 studies, excluding results for "rural" settings). Ranges of reported central value statistics and disaggregate (sample-level) values are presented, including the country where the low and high measurements were taken.

Table 1. Summary of the 42 studies directly measuring on-road bicyclists' exposure concentrations

Pollutant	Units [1]	# studies (N) [2]	Years of studies	Reported Central Values [3]				Reported Disaggregate Values	
				Mean (SD)	Median	Low	High	Low	High
CO	ppm	12 (16)	1976-2013	2.8 (3.9)	1.2	0.5 (New Zealand)	13 (USA)	0.1 (New Zealand)	21 (USA)
NO_2	μg/m^3	4 (5)	1995-2006	55.8 (33.8)	46.3	26 (Australia)	114 (Netherlands)	8 (Australia)	262 (Netherlands)
VOC [4]		9	1991-2011						
benzene	μg/m^3	9 (12)	1991-2011	17.2 (18.3)	10.6	0.34 (Canada)	56 (UK)	0.1 (Canada)	120 (UK)
toluene	μg/m^3	6 (7)	1991-2011	57.6 (51.6)	50.5	1.07 (Canada)	122 (UK)	0.3 (Canada)	1,230 (Italy)
xylenes	μg/m^3	5 (6)	1991-2011	48.9 (45.2)	44.3	0.6 (Canada)	105 (Italy)	0.15 (Canada)	281 (Italy)
PM [5]									
UFP, PNC	pt/cc	18 (31)	2005-2013	28,450 (18,169)	24,800	8,734 (Belgium)	93,968 (UK)	1,900 (USA)	1,033,188 (USA)
$PM_{2.5}$	ug/m3	17 (29)	2001-2014	29.9 (22.8)	23.5	4.88 (USA)	88.1 (Ireland)	0 (Netherlands)	130 (UK)
PM_{10}	ug/m3	10 (15)	2001-2014	50.2 (12.0)	50.0	32.0 (New Zealand)	72.7 (Belgium)	8.2 (New Zealand)	160 (Belgium)
BC, EC [6]	ug/m3	9 (14)	2002-2013	6.85 (7.09)	3.04	1.05 (Canada)	21.0 (UK)	0.09 (USA)	63.83 (USA)

[1] Conversion of reported values between μg/m^3 and ppb or ppm assumes molar gas volume of 24.45 L
[2] Some studies report separate central value results by route. All routes are included except those designated as "rural" settings. See Supplemental Material for details.
[3] When multiple central value statistics are reported in a study, a single value was selected as the arithmetic mean, geometric mean, or median, in that order
[4] Various compounds are reported in the studies measuring VOC; only benzene, toluene, and (o-, m-, and p-) xylenes are reported in more than half of the VOC studies
[5] In addition to BC, UFP, $PM_{2.5}$, and PM_{10}, 6 studies report PM of other sizes (PM_1 through PM_5) over the years 1991 to 2013.
[6] Excludes three additional studies that only report BC concentrations in units of absorbance

The mean on-road measurements in Table 1 are all well above typical urban background concentrations (see Supplemental Material). Table 1 shows that measured bicyclist exposure concentrations for most pollutants exhibit high variability among studies, with a standard deviation (SD) greater than 50% of the mean value for all pollutants except PM_{10}, and a SD greater than the mean for CO, benzene, and BC/EC. Bicyclists' average CO exposure concentrations have been measured in the range of 0.5 to 13 ppm, though all studies after 1995 report central value concentrations below 3 ppm.

3.1 Modal Comparisons of Exposure Concentration

A popular study design for traveler exposure studies is modal comparisons, in which exposure concentrations are compared for travelers using different transportation modes between the same origin and destination or along identical or parallel routes. Results from modal comparisons of exposure are inconsistent. Bicyclists sometimes have lower exposure concentrations than motorized modes, especially when they use facilities that are separated from traffic (H. S. Adams, Nieuwenhuijsen, & Colvile, 2001; H. S. Adams, Nieuwenhuijsen, Colvile, Older, & Kendall, 2002; Boogaard, Borgman, Kamminga, & Hoek, 2009; Chertok, Voukelatos, Sheppeard, & Rissel, 2004; de Nazelle et al., 2012; Dons et al., 2012; Kaur et al., 2007; Kingham, Longley, Salmond, Pattinson, & Shrestha, 2013; Kingham, Meaton, Sheard, & Lawrenson, 1998; Knibbs et al., 2011; McNabola, Broderick, & Gill, 2008; van Wijnen, Verhoeff, Jans, & Bruggen, 1995). But modal comparison studies have also found insignificant differences in concentrations by mode, significantly higher bicyclist exposure concentrations than other modes, or inconsistent results by pollutant, location, or time of day (Boogaard et al., 2009; Chertok et al., 2004; de Nazelle et al., 2012; Int Panis et al., 2010; Kaur & Nieuwenhuijsen, 2009; Kingham et al., 2013; Nwokoro et al., 2012; Quiros, Lee, Wang, & Zhu, 2013; Ragettli et al., 2013; Waldman et al., 1977; Yu et al., 2012). Likely causes of inconsistent results across studies include differences in the proximity and intensity of motor vehicle traffic, varying availability and use of bicycle facilities, and instrumentation/sampling differences (see Supplemental Material for information on study methods).

Modal comparison exposure studies typically use the same routes or origins and destinations across modes and fix other travel characteristics (e.g. departure time). While potentially informative, these comparisons are not always realistic because pollution exposure is also affected by intrinsic modal travel differences. The more realistic modal comparisons allow self-selected routes or direct active travelers to use representative routes for their mode – but local transportation network characteristics may affect the results. Bicycle travel patterns are different from motorized ones because of distinct traveler characteristics, trip distances, and route preferences (Broach, Dill, & Gliebe, 2012; Plaut, 2005). Real-world bicycle trips tend to be shorter and in higher-density parts of a city than trips using motorized modes. Bicycle trips are also highly seasonal (Nankervis, 1999), so a different distribution of meteorological conditions could be expected by mode, with a systematic influence on exposure concentrations. Most bicycle exposure studies occur during warmer months when a greater proportion of bicycling occurs (see the Supplemental Material), but the joint seasonality of mode splits and pollution levels should be considered when comparing travelers' exposures – especially for year-round bicyclists.

Although modal comparisons can be informative, they rarely provide practical insights into how to reduce exposure concentrations, other than mode shifts. Modal comparison studies rarely vary within-mode factors (such as route choice), which can be the most important determinants of exposure concentrations during travel (Knibbs et al., 2011).

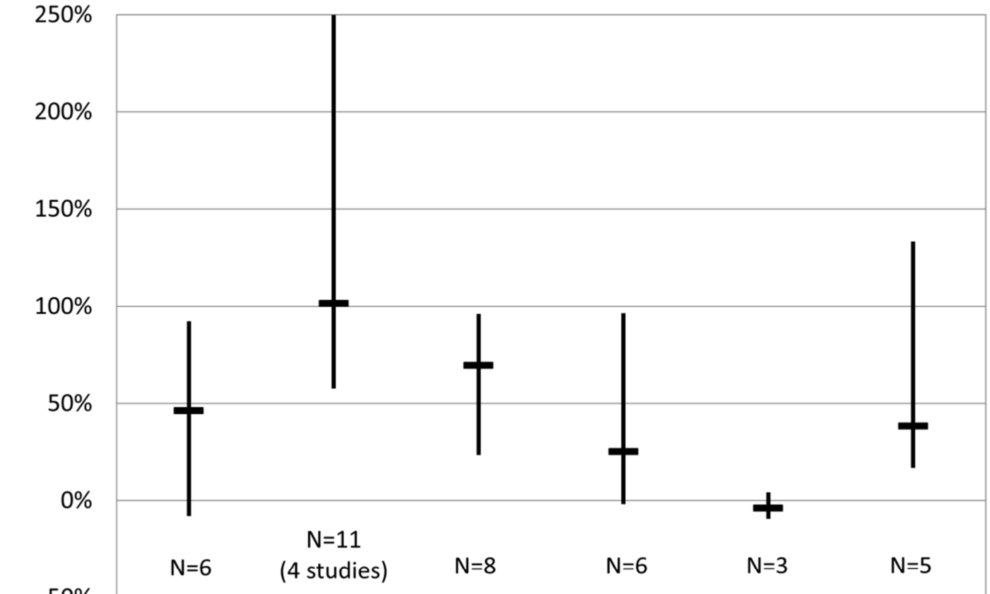
3.2 Factors Affecting Bicyclists' Exposure Concentrations

Multivariate analyses of travelers' exposure concentrations have shown that important factors include wind and weather, traffic and route, and the built environment around the roadway (H. S. Adams et al., 2001; Berghmans et al., 2009; Boogaard et al., 2009; Hatzopoulou et al., 2013; Kaur et al., 2007; Kaur & Nieuwenhuijsen, 2009; Knibbs et al., 2011; McNabola, Broderick, & Gill, 2009a; Quiros et al., 2013). But few studies have looked at bicyclist-specific factors that could influence exposure, such as lateral position in the road, proximity to exhaust pipes, breathing height, and the ability to "dodge between" vehicles (Kaur et al., 2007).

Wind is consistently a significant factor for exposure, decreasing concentrations through dispersion (H. S. Adams et al., 2001; Hatzopoulou et al., 2013; Hong & Bae, 2012; Jarjour et al., 2013; Kaur et al., 2007; Kaur & Nieuwenhuijsen, 2009; Kingham et al., 1998; Knibbs et al., 2011; McNabola et al., 2009a). Temperature is less consistently a significant factor, and effects can be difficult to distinguish from humidity because of a strong negative correlation (H. S. Adams et al., 2001; Hatzopoulou et al., 2013; Kaur et al., 2007; Kaur & Nieuwenhuijsen, 2009; Kingham et al., 1998; Knibbs et al., 2011). Time-of-day is a factor that incorporates influencing effects of local weather and diurnal traffic patterns – particularly relevant for urban areas with diurnal temperature inversions that significantly affect pollutant levels.

After weather, the next most important factors for bicyclists' exposure concentrations can be combined into a single category: separation from motor vehicle traffic. These factors include the concentration-reducing effects of traveling on low-traffic routes (Hatzopoulou et al., 2013; Hertel, Hvidberg, Ketzel, Storm, & Stausgaard, 2008), on separated bicycle facilities (Hatzopoulou et al., 2013; Hong & Bae, 2012; Kendrick et al., 2011; Kingham et al., 2013, 1998), and during off-peak periods or weekends (Dons et al., 2013; Huang et al., 2012; Kleiner & Spengler, 1976). Lacking more specific data, the influence of motor vehicle traffic on exposure concentrations is sometimes estimated using a proxy of facility type, time-of-day, or average daily traffic (ADT) estimates (Boogaard et al., 2009; Cole-Hunter, Morawska, Stewart, Jayaratne, & Solomon, 2012; Hong & Bae, 2012; Jarjour et al., 2013; Ragettli et al., 2013; Weichenthal et al., 2011).

The influence of motor vehicle traffic was measured in 14 different studies by comparing bicyclists' exposure concentrations on "high traffic" and "low traffic" routes or using a related dichotomy (inner-city/suburban, on-road/off-road, near-road/cycle path). The combined results are shown in Figure 2, with the median and range of reported percent increases on "high traffic" versus "low traffic" routes (see the Supplemental Material for sources). As expected, pollutants that are more dominated by motor vehicle sources in roadway environments (hydrocarbon VOC, UFP) show larger increases on high-traffic routes.



	CO	VOC	UFP	PM2.5	PM10	BC
High	92%	250%	96%	97%	4%	134%
Low	-8%	58%	24%	-2%	-9%	17%
– Median	47%	102%	70%	26%	-4%	39%

Figure 2. Reported Increases in Bicyclists' Exposure Concentrations in "High Traffic" versus "Low Traffic" Routes and Locations*

* Urban/rural comparisons are excluded. Where multiple observations are reported per study (e.g. by city or time period), a weighted average by number of samples was used. For VOC, reported BTEX compounds (benzene, toluene, ethylbenzene, and xylenes) are included (11 comparisons for these compounds in 4 different studies). Sources: CO (Bevan et al., 1991; Jarjour et al., 2013; Kingham et al., 2013; Kleiner and Spengler, 1976; Waldman et al., 1977; Weichenthal et al., 2011), VOC: (Bevan et al., 1991; Kingham et al., 1998; McNabola et al., 2008; Weichenthal et al., 2011), UFP: (Cole-Hunter et al., 2013, 2012; Jarjour et al., 2013; Kingham et al., 2013; Ragettli et al., 2013; Strak et al., 2010; Weichenthal et al., 2011; Zuurbier et al., 2010), $PM_{2.5}$: (Adams et al., 2001; Jarjour et al., 2013; Kingham et al., 2013; McNabola et al., 2008; Weichenthal et al., 2011; Zuurbier et al., 2010), PM_{10}: (Kingham et al., 2013; Strak et al., 2010; Zuurbier et al., 2010), BC: (Jarjour et al., 2013; Kingham et al., 1998; Strak et al., 2010; Weichenthal et al., 2011; Zuurbier et al., 2010)

Explicit traffic variables such as motor vehicle volume or speed are often not included in bicyclist pollution exposure analysis because of a lack of concomitant data. When assessed, vehicle volumes, particularly truck or diesel vehicles, generally have a positive influence on pollutant exposure concentrations, though they are not always significant variables (Boogaard et al., 2009; Dons et al., 2013; Hatzopoulou et al., 2013; Kaur & Nieuwenhuijsen, 2009; Knibbs et al., 2011; McNabola et al., 2009a; Quiros et al., 2013). Aggregate traffic variables such as ADT cannot reveal the potentially important influences of varying traffic volumes, speeds, queuing, and fleet composition over the data collection periods. Furthermore, highly aggregate traffic variables are often correlated with geometric roadway characteristics such as the number of lanes, which also influence pollutant concentrations through dispersion.

Traffic data used in bicycle exposure studies to date have been non-specific to the study period, limited in spatial and temporal coverage, and/or highly aggregated (in time and vehicle type). Of the 42 studies included in Table 1, only 4 report traffic data collected at the locations and time periods of air quality measurements. Kaur et al. (2005)[1] and McNabola et al. (2008)[2] retrieved unclassified hourly vehicle volumes from traffic signal data at major intersections on the study routes. Hatzopoulou et al. (2013) collected intermittent manual vehicle counts using 5 vehicle classes for 10-20 minute periods sequentially at dozens of locations around the on-road measurement area. Quiros et al. (2013) performed intermittent manual vehicle counts for 5-minute periods using 9 vehicle classes (including bicycles and pedestrians) at a single location on the study corridor.

The next major factors for exposure concentrations, after weather and motor vehicle traffic, are the study setting and methodology. Comparing measured exposure concentrations across studies reveals wide ranges (Table 1), indicative of different study settings (time frame, city, locational characteristics) and different experimental methods (instruments, sampling strategy, aggregation, etc.). Potentially important differences among study settings include traffic patterns, weather conditions, vehicle fleets and fuels, urban form, and topography. Boogaard et al. (2009) compare bicyclists' on-road exposure concentrations in 11 Dutch cities over a 3-month period (using a consistent methodology) and report coefficients of variability for UFP and $PM_{2.5}$ of 0.22 and 0.86 among cities. For comparison, the coefficients of variability for UFP and $PM_{2.5}$ among studies in Table 1 are 0.64 and 0.76.

4 Bicyclists' Air Pollution Intake

The mass of air pollutants that cross the body boundary through the mouth and nose is the intake dose (Ott et al., 2007). Estimates of intake dose rates per unit time combine exposure concentrations with a respiration rate; intake dose rates per unit distance also take travel duration into account (as does total intake dose over a journey). Some studies consider only duration (not respiration) by estimating cumulative exposure, such as (Nwokoro et al., 2012; Ragettli et al., 2013). Measurement and analysis of bicyclists' pollutant intake facilitates a transition toward a dose-oriented estimation of health effects.

4.1 Respiration

Respiration rate is commonly expressed as the minute respiratory volume (or minute ventilation, V_E) – which is the volume of air displaced per minute. Minute respiratory volume is the product of the tidal volume V_T and the breathing frequency f_r (breaths per minute). Tidal volume V_T is the volume of air displaced in a single breath; typical ranges are 1.4 to 2.2 liters (L) for bicyclists and 0.6 to 0.8 L for persons at rest or in a car (Int Panis et al., 2010). Multiplying

[1] Traffic data are reported in a companion paper, Kaur and Nieuwenhuijsen (2009).
[2] Traffic data are only used in a companion paper, McNabola et al. (2009b).

V_E by the average exposure concentration yields the average pollutant inhalation rate in mass per unit time.

Table 2 summarizes published traveling bicyclists' respiration parameters (see Supplemental Material for a description of the measurement methods). Minute ventilation has been reported as 22 to 59 L/min for bicyclists: 2 to 5 times higher than for travelers in automobiles or at rest. Bernmark et al. (2006) found V_E peaks for bicycle messengers of up to 97 L/min. The ranges of minute ventilations in Table 2 are related to the different average travel speeds and heart rates among the studies (included in Table 2), as well as potentially other experimental differences such as terrain, bicycle weight and condition, weather, and subject fitness. Greater exertion increases V_E primarily by an increase in V_T at lower levels of exercise and by an increase in f_r at higher levels of exercise; f_r is the dominant factor at 70-80% of peak exercise level (Weisman, 2003). Trained professional bicyclists can achieve a greater increase in V_E through increases in V_T than recreational bicyclists (Faria, Parker, & Faria, 2005b).

Table 2. Respiration-Related Parameters Measured for Bicyclists

Group	Minute ventilation, V_E (L/min)	Tidal volume (L)	Breathing frequency (min^{-1})	Heart rate (bpm)	Speed (kph)	Ratio of bicycle/car V_E [1]	Reference & Method [2]
All	23.5			100	12	2.0	1, estimated
	28.7				13.5	2.5	2, on-road
Male	22			94	12	1.8	1, estimated
	22.7				14	1.9	3, on-road
	25	1.25	20		8	2.1	4, lab
	28					2.3	5, lab
	31			107		2.6	6, estimated
	31.4				19.5	2.6	3, on-road
	44.2			138	20	3.7	7, estimated
	50	1.92	26		19	4.2	4, lab
	51.2				24	4.3	3, on-road
	59.1	2.2	27.9	129.6	20.5	4.9	8, on-road
Female	22.6				14	2.1	3, on-road
	27.6			116	12	2.5	1, estimated
	32.8				19.5	3.0	3, on-road
	46.2	1.4	32.7	140	19.5	4.2	8, on-road
	51.8				24	4.7	3, on-road

Blank cells are not reported

[1] Reference minute ventilation for car drivers of 12 L/min for Males, 11 L/min for Females, and 11.5 L/min for All, based on (Adams, 1993; Int Panis et al., 2010; O'Donoghue et al., 2007; van Wijnen et al., 1995; Zuurbier et al., 2009)

[2] References: 1 (Zuurbier et al., 2009), 2 (van Wijnen et al., 1995), 3 (Adams, 1993), 4 (McNabola et al., 2007), 5 (O'Donoghue et al., 2007), 6 (Bernmark et al., 2006), 7 (Cole-Hunter et al., 2012), 8 (Int Panis et al., 2010) Methodologies are categorized as: "on-road" (direct on-road measurement of respiration using masks), "lab" (laboratory ergometer-based respiration measurements), and "estimated" (on-road measurement of heart rate and estimation of respiration using laboratory ergometer-based heart rate/ventilation relationships)

For active travelers such as bicyclists, V_E will be a function of travel characteristics that determine power requirements. The major determinants of power output during bicycling are energy losses (resistance) and changes in kinetic and potential energy (acceleration and grades, respectively). The largest energy losses are typically aerodynamic drag followed by rolling resistance. Rolling resistance becomes a more important factor at lower speeds and in still air, when drag is less severe (di Prampero, Cortili, Mognoni, & Saibene, 1979; Faria et al., 2005a; Martin, Milliken, Cobb, McFadden, & Coggan, 1998; Olds, 2001; Whitt, 1971; Wilson, 2004). Nadeau et al. (2006) measured V_E of around 12, 23, and 35 L/min for bicycle ergometer workloads of 0, 50, and 100 W, respectively – suggesting that the subjects in the studies in Table 2 experienced workloads ranging from around 50 W to well over 100 W of power.

Compilations of physical activity data often use MET units to compare energy expenditure with a standardized unit; a MET is defined as $\text{MET} = \frac{\dot{e}}{\text{RMR}}$ where $\dot{e}$ is the rate of metabolic energy production and RMR is the resting metabolic rate (Ainsworth et al., 2011a, 2011b; U.S. Environmental Protection Agency, 2009). RMR is an individual-specific value (varying across individuals), often assumed to be 3.5 ml-O_2/min per kg body mass – i.e. $\text{MET} = \frac{\dot{e}}{K \cdot m}$, where K is a constant and m is body mass. Thus, MET values are directly proportional to energy expenditure for an individual and inversely proportional to an individual's body mass for a given energy expenditure[3].

Resting activities are at a MET of 1, while "general" bicycling is at a MET of 7.5 and bicycling "to/from work, self selected pace" is at MET 6.8 in the "Compendium of Physical Activities" (Ainsworth et al., 2011a, 2011b). The Compendium lists 16 different types of bicycling as activities with energy expenditures ranging from 3.5 MET for "leisure" bicycling at 5.5 mph to 16 MET for competitive mountain bicycle racing. Non-sport bicycling has been estimated to require 3.5 to 9 MET of energy expenditure, with power output of roughly 50 to 150 W, depending on the speed (Bernmark et al., 2006; de Geus, de Smet, Nijs, & Meeusen, 2007; Whitt, 1971). MET values have been employed to estimate bicyclists' respiration for pollution dose assessments using both reference MET values and MET values estimated from accelerometer measurements; average accelerometer-based MET for bicycling was estimated at 6.58 with a corresponding ventilation rate of 41 L/min (de Nazelle et al., 2012). Respiration was estimated from MET values using stochastic relationships between oxygen uptake rates and ventilation rates along with the individuals' body mass (de Nazelle, Rodríguez, & Crawford-Brown, 2009; Johnson, 2002).

4.2 Studies of Bicyclists' Pollution Intake

Table 3 characterizes published studies of bicyclists' air pollution exposure, intake, uptake, or biomarkers that use spatially-explicit exposure concentration data (modeled or

[3] It should be noted that metabolic energy expenditure during bicycling is the sum of energy expenditure for baseline functions and the rate of external work (Olds, 2001). Assuming that the baseline energy expenditure is roughly equal to the RMR, the MET can be expressed as a function of external power output p as $\text{MET} = 1 + \frac{p}{RMR}$. Thus, MET values increase linearly (but not proportionally) with the *external* power demands of bicycling.

measured). Studies are categorized according to how (and whether) they account for 1) respiration (i.e. intake), 2) uptake of gases or deposition of particles, and 3) health biomarkers. The last two dimensions are discussed in Sections 5 and 6, respectively. “Constant” respiration refers to studies that apply fixed respiration rates by mode or individual; “variable” respiration refers to studies that use varying respiration rates by trip or at a greater level of detail. The categorization in Table 3 proceeds roughly from least to most comprehensive (A to M) in terms of targeting farther along the exposure-health pathway, assessing linkages more directly (e.g. measuring versus assuming), and/or examining more intermediate steps between exposure and uptake or biomarkers.

Table 3. Categorization of Bicyclists' Air Pollution Exposure, Intake, Uptake, and Biomarker Studies

		1) Respiration [2]						2) Uptake/Deposition			3) Biomarkers
		Constant				Variable					
Study Type	Number of Studies [1]	Assumed	Modeled	Measured in-lab	Measured on-road	Modeled	Measured on-road	Assumed	Modeled	Measured	Measured
A	28										
B	7	●									
C	3		●								
D	1			●							
E	1				●						
F	1					●					
G	1		●					●			
H	1			●					●		
I	1						●		●		
J	2									●	
K	8										●
L	2		●								●
M	1					●			●		●
Totals:	57	7	6	2	1	2	1	1	3	2	11

Grey cells mean that dimension was not assessed (respiration, uptake, biomarkers); ● indicates the method of assessment for that dimension by each study type

[1] Includes all published papers of on-road bicyclists' pollution exposure with spatially-explicit exposure concentration data

[2] "Constant" respiration means fixed respiration rates by mode or individual; "variable" respiration means varying respiration by trip or greater level of detail

A: (Adams et al., 2002, 2001a, 2001b; Bean et al., 2011; Berghmans et al., 2009; Bevan et al., 1991; Boogaard et al., 2009; Chan et al., 1994; Chertok et al., 2004; Dekoninck et al., 2013; Dons et al., 2013; Farrar et al., 2001; Gee and Raper, 1999; Hatzopoulou et al., 2013a, 2013b; Hertel et al., 2008; Hong and Bae, 2012; Kaur and Nieuwenhuijsen, 2009; Kaur et al., 2005; Kendrick et al., 2011; Kingham et al., 2013, 1998; Kleiner and Spengler, 1976; McNabola et al., 2009; Ragettli et al., 2013; Sitzmann et al., 1999; Strauss et al., 2012; Thai et al., 2008), **B**: (Dirks et al., 2012; Dons et al., 2012; Fajardo and Rojas, 2012; Huang et al., 2012; Quiros et al., 2013; Rank et al., 2001; Yu et al., 2012), **C**: (Bernmark et al., 2006; de Nazelle et al., 2012; Zuurbier et al., 2010), **D**: (O'Donoghue et al., 2007), **E**: (van Wijnen et al., 1995), **F**: (Cole-Hunter et al., 2012), **G**: (Vinzents et al., 2005), **H**: (McNabola et al., 2008), **I**: (Int Panis et al., 2010), **J**: (Bergamaschi et al., 1999; Nwokoro et al., 2012), **K**: (Bos et al., 2011; Cole-Hunter et al., 2013; Jacobs et al., 2010; Jarjour et al., 2013; Strak et al., 2010; Waldman et al., 1977; Weichenthal et al., 2012, 2011), **L**: (Zuurbier et al., 2011a, 2011b), **M**: (Nyhan et al., 2014)

Many studies consider only exposure concentrations and neglect the question of intake dose and the issue of varying respiration and energy expenditure by travel mode and condition (Type A). Similarly, some studies measure exposure concentrations and uptake doses or health biomarkers directly, but do not address the intermediate step of intake or respiration (Types J and K). Of the 19 studies in Table 3 that explicitly consider respiration, 16 use fixed values of V_E for each travel mode or individual (Types B-E, G, H, and L). Type B studies (7 of the 19) apply an assumed V_E for bicyclists based on other published research. Two studies (Types D and H) use bicycle ergometers in a laboratory to determine representative respiration values by mode. Of the 8 studies that model respiration (Types C, F, G, L, and M), 6 use ergometers to develop individual subject functions to estimate on-road V_E from field-measured HR, 1 uses previously-developed V_E-HR functions with field-measured HR, 1 estimates respiration from accelerometer-based MET values – see Section 4.1. Only 2 of these 8 studies (Types F and M) estimate intake using variable ventilation rates by trip (Cole-Hunter et al., 2012) or at 2-minute aggregations (Nyhan, McNabola, & Misstear, 2014).

Two studies in Table 3 directly measure on-road bicyclists' minute ventilation in order to estimate intake dose (Types E and I). Van Wijnen et al. (1995) use fixed mode-specific respiration rates that are the averages of measured on-road minute ventilation for a set of test subjects traveling on the same test routes as the concentration measurements, but at different times. Int Panis et al. (2010) use simultaneously monitored on-road respiration and concentration data to estimate intake dose. Combining tidal volume and pollutant concentration measurements, Int Panis et al. calculate breath-by-breath mass intake and sum over trips, thus including both respiration and duration effects on total intake.

Table 3 shows that there has been little assessment of the variability of bicyclists' respiration as they travel in an urban environment. If the variability in respiration is independent of exposure concentrations, then representative averages for each will suffice (assuming linearity). But there is likely to be spatial correlation between pollutant concentrations and bicyclist energy expenditure at locations such as intersections and hills, where both motor vehicles and bicyclists are required to generate more energy. There is also a potential correlation between exposure duration and exposure concentration at congested bottlenecks or busy intersections. At the route level, Cole-Hunter et al. (2013) found no significant differences in measured HR for routes with low and high proximity to traffic; they conclude that variability in UFP intake dose for bicyclists would be predominantly determined by exposure concentrations, not ventilation characteristics. But a wide range of bicyclists' respiration values have been reported (Section 4.1), and the lack of bicyclist intake dose studies considering variable respiration rates leaves the question open.

4.3 Modal Comparisons of Pollution Intake

Int Panis (2010) argues that comparisons of exposure concentrations by travel mode (as in Section 3.1) are "not entirely relevant" because of the dominating effect of breathing differences among modes. Modal comparisons of pollution intake dose go beyond exposure concentrations by including respiration to compare intake dose rates per unit time. More detailed comparisons also consider the intake effects of travel duration differences, assessing intake doses per trip or unit travel distance. For faster trips, the time spent in an alternative environment is typically neglected; this aspect may be important when the air quality at the trip destination is poor. Inherent speed differences by mode are problematic for modal comparisons of intake rates by either normalization.

Table 4 summarizes the 12 published modal comparisons that include respiration, showing the median and range for ratios of bicycle to alternative mode intake or uptake doses. Dose ratios are presented separately for the 8 studies that compare doses per unit distance and the 5 studies that compare doses per unit time (1 assesses both). For most pollutants, studies that compare doses per unit distance find greater bicycle/car dose ratios than comparisons per unit time, as expected from bicyclists' lower travel speeds. This body of literature is still much smaller than modal comparisons of exposure, but for the most part 2 to 5 times higher ventilation rates and slower travel speeds for bicyclists compared to motor vehicle passengers outweigh any beneficial exposure concentration differences. Bicyclists' doses are less consistent when compared to pedestrians, which is not surprising because walking is another active travel mode with elevated respiration. Pedestrians typically have lower respiration rates (McNabola et al., 2007) but also lower speeds, with counteracting effects on intake rates per unit distance.

Table 4. Ratios of Intake or Uptake Doses for Bicyclists versus Other Modes

		Per unit distance [1]		Per unit time	
	Alternative Mode	N [2]	Median (Range)	N	Median (Range)
CO	Pedestrian	1	0.80	0	
	Car/Taxi	3	1.09 (0.36-4.67)	1	0.87
	Bus	3	1.63 (1.07-4.67)	0	
	Rail	1	7.00	0	
VOC [3]	Pedestrian	1	1.11	0	
	Car	1	0.81	4 (2 studies)	0.71 (0.50-0.72)
	Bus	2	1.60 (1.25-1.96)	0	
NO_2	Car	0		1	3.08
UFP	Pedestrian	2	0.68 (0.51-0.84)	0	
	Car	3	5.42 (1.00-10.42)	1	2.09
	Bus	1	1.90	1	1.87
$PM_{2.5}$	Pedestrian	4	1.13 (0.47-1.97)	1	2.09
	Car/Taxi	5	3.36 (1.38-10.88)	1	1.70
	Bus	4	1.77 (1.06-4.78)	2	3.14 (1.91-4.36)
	Rail	1	2.56	1	2.29
PM_{10}	Pedestrian	1	1.62	1	1.82
	Car	1	6.75	1	1.66
	Bus	1	3.21	2	2.13 (1.15-3.10)
	Rail	1	3.06	1	2.21
BC	Pedestrian	1	0.81	0	
	Car	1	0.84	2	1.90 (1.36-2.44)
	Bus	1	1.64	1	1.51

[1] Values are ratios of bicycle to alternative mode doses in mass, particles, or ppb per unit distance (i.e. per km or per trip) or per unit time (i.e. per hour of travel); the table includes all studies that directly compare pollutant intake or uptake between travelers by bicycle and other modes for similar trips.
[2] A single mean value (weighted by number of samples) was computed for studies reporting separate results by routes or times of day. VOC doses per unit time are from 2 studies, with one reporting 3 different compounds.
[3] Only reported values for BTEX compounds are included.

Sources, per unit distance: **CO**: (de Nazelle et al., 2012; Dirks et al., 2012; Huang et al., 2012), **VOC**: (McNabola et al., 2008; O'Donoghue et al., 2007), **UFP**: (de Nazelle et al., 2012; Int Panis et al., 2010; Quiros et al., 2013), **$PM_{2.5}$**: (de Nazelle et al., 2012; Huang et al., 2012; Int Panis et al., 2010; McNabola et al., 2008; Nyhan et al., 2014; Quiros et al., 2013), **PM_{10}**: (Int Panis et al., 2010; Nyhan et al., 2014), **BC**: (de Nazelle et al., 2012)
Sources, per unit time: **CO**: (van Wijnen et al., 1995), **VOC**: (Rank et al., 2001; van Wijnen et al., 1995), **NO_2**: (van Wijnen et al., 1995), **UFP**: (Zuurbier et al., 2010), **$PM_{2.5}$**: (Nyhan et al., 2014; Zuurbier et al., 2010), **PM_{10}**: (Nyhan et al., 2014; Zuurbier et al., 2010), **BC**: (Dons et al., 2012; Zuurbier et al., 2010)

Few of the modal comparisons of dose directly measure on-road respiration or model respiration as a function of travel characteristics beyond mode. This is important because travel attributes such as road grade and speed affect respiration and inhalation rates for bicyclists but not motorized modes. Intake doses per trip will be further affected by duration changes with route and destination choices, which are normally not varied in modal comparisons (as discussed in Section 3.1). Furthermore, active travelers tend to have unique demographics (Plaut, 2005), which could systematically impact respiration through physiological attributes such as sex and health condition (W. C. Adams, 1993).

5 Bicyclists' Air Pollution Uptake

A portion of inhaled pollutants are either absorbed (gases) or deposited (particles) onto the lining of the respiratory tract or into the bloodstream. Absorbed/deposited pollutants are then either expelled (through mucociliary clearance or desorption) or transported to body tissues. The air pollution uptake dose is the amount of pollutant that is not exhaled or expelled, but rather incorporated into the body (Figure 1).

Table 5 summarizes the factors that are expected to increase pollutant uptake for bicyclists. The first two factors reflect the exposure in terms of concentration and duration. The next set of factors in Table 5 is attributes of the pollutants that determine uptake dose (independent of travel characteristics). Particle size is important for PM uptake because deposition and clearance rates vary with particle size. UFP deposition is also influenced by the particles' growth characteristics in high humidity conditions such as in lung airways (hygroscopicity). Gas reactivity and solubility in blood and lipids are similarly important because they affect absorption and diffusion rates (Daigle et al., 2003; International Commission on Radiological Protection [ICRP], 1994; Löndahl et al., 2007; McNabola et al., 2008; Ott et al., 2007; West, 2012).

Table 5. Factors that Increase Pollutant Uptake

Factor	Increased uptake with:
Exposure	
Concentration	Higher concentrations
Duration	Longer duration
Pollutant	
Particle size	Smaller particles
Particle hygroscopicity	More hydrophobic particles
Gas solubility	More blood- and lipid-soluble compounds
Respiration/physiology	
Breath volume flow rate (V_E)	Greater ventilation
Depth of breathing (V_T)	Greater tidal volume
Path of breathing	Oral breathing
Cardiac output (lung perfusion)	Greater perfusion
Metabolic rate	Higher metabolic rate

Table 5 also summarizes the physiology and respiration factors that influence uptake. Intake dose is determined by V_E and the exposure concentration; uptake dose is further influenced by the depth of respiration (V_T) and the amount of oral breathing. Greater uptake fractions of inhaled PM occur during deeper and more oral breathing (ICRP, 1994), which are associated with higher levels of exertion (Samet et al., 1993; Weisman, 2003). Daigle et al. (2003) found that when subjects' V_E increased from 11.5 to 38.1 L/min the deposition fraction (DF), the portion of particles that are not exhaled after inhalation, increased from 0.66 to 0.83 by number of particles and from 0.58 to 0.76 by mass of particles. Thus, a V_E increase by a factor of 3.3 led to a total deposition increase by a factor of 4.5 due to a higher DF. Löndahl et al. (2007) found only small changes in DF for UFP (by less than 0.03) during exercise when compared to rest (V_E of 33.9 versus 7.8 L/min), but both of these studies found that established models under-predicted deposition of UFP – especially during exercise.

Uptake rates for gaseous pollutants are also affected by the characteristics of the gas and the level of physical exertion. VOC and CO uptake rates are several times greater during exercise than at rest for a given exposure concentration. But the uptake *fraction* of inhaled gases tends to decrease with exertion level because gas uptake rates increase more slowly than intake rates with exercise. (Astrand, Engstrom, & Ovrum, 1978; Astrand, 1985; Filley, MacIntosh, & Wright, 1954; Nadeau et al., 2006; Pezzagno, Imbriani, Ghittori, & Capodaglio, 1988). Diffusion-limited gases such as CO are primarily impacted by the diffusing capacity of the lungs, which can increase by a factor of three during exercise (West, 2012). Uptake rates for perfusion-limited gases such as low-solubility VOC and NO_2 increase with ventilation and perfusion of the lungs, gas partial pressure differences between blood and air, and gas solubility in blood (Astrand, 1985; Csanády & Filser, 2001; Farhi, 1967; West, 2012). As blood concentrations approach equilibrium with inspired air, the uptake rate will fall to the steady-state rate of metabolic clearance (Csanády & Filser, 2001; Wallace, Pellizzari, & Gordon, 1993). Although exercise increases ventilation and perfusion, it also can decrease the rate at which pollutants are metabolized by reducing blood flow to the liver – reducing the steady-state uptake rate while simultaneously increasing blood concentrations (Astrand, 1985; Csanády & Filser, 2001; Kumagai & Matsunaga, 2000; Nadeau et al., 2006).

Detailed uptake models allow estimation of different locations/tissues of pollutant uptake, which is relevant because of varying susceptibility to negative health effects from air pollution uptake by different tissues. Common uptake models include body compartment and physiologically based pharmacokinetic (PBPK) models for gases and human respiratory tract models for both gases and PM (Heinrich-Ramm et al., 2000; Hofmann, 2011; ICRP, 1994; King et al., 2011; Ott et al., 2007; Wallace, Nelson, Pellizzari, & Raymer, 1997; Wallace et al., 1993). Uptake models are generally validated using much steadier air concentrations than have been observed in on-road environments, so it is not clear how applicable they are for on-road uptake analysis with highly transient exposure concentrations.

Uptake of air pollutants by bicyclists has been studied less than exposure concentrations or intake doses (6 of the 57 studies in Table 3 explicitly consider uptake). Vinzents et al. (2005)

conservatively estimate deposition as linearly proportional to workload (on average 43% higher deposition of PM while bicycling than at rest). Int Panis et al. (2010) use DF that vary with V_E, V_T, and particle size, based on two previous studies of particle deposition (Chalupa, Morrow, Oberdorster, Utell, & Frampton, 2004; Daigle et al., 2003). Although other factors in Table 5 were not explicitly modeled, these reference studies used physically active subjects and traffic exhaust particles. Intake doses of UFP were 4.2 to 6.6 times higher for bicyclists than car passengers, while uptake doses were 5.1 to 8.3 times higher – despite lower or roughly equivalent exposure concentrations for bicyclists. $PM_{2.5}$ comparisons were similar, with intake doses 5.7 to 7.6 times higher for bicyclists than car passengers but uptake doses 8.0-12.0 times higher.

McNabola et al. (2008) modeled uptake of VOC and $PM_{2.5}$ using the ICRP human respiratory tract model (ICRP,1994) with on-road measured exposure concentrations and laboratory-measured respiration characteristics for bicycle, pedestrian, car, and bus modes. The ICRP model can include all relevant factors in Table 5 except lung perfusion, though the assumed fraction of oral breathing is not reported by McNabola et al.. Bicyclists had the highest total lung deposition of $PM_{2.5}$ and the second-highest absorption of VOC over similar trips to other modes. Breathing characteristics (frequency, tidal volume) and VOC solubility affected the uptake dose and the location of absorption, with more benzene absorbed deep in the lungs for bicyclists and pedestrians. Breathing differences also affected benzene absorption more than 1,3-butadiene absorption because of benzene's lower solubility. McNabola et al. (2007) similarly model VOC uptake by bicyclists using different travel speeds, but with assumed (rather than measured) exposure concentrations. They found that higher bicycling speeds reduce VOC absorption over a fixed travel distance because the increase in respiration rate is smaller than the reduction in exposure duration.

The same ICRP model was also applied by Nyhan et al. (2014) to estimate $PM_{2.5}$ and PM_{10} lung deposition for trips by bicycle, foot, bus, and train. Their estimates indicate that bicyclists' PM intake and uptake per trip is disproportionately higher than exposure concentrations compared to other modes. But the cross-mode ratios are equivalent for modeled intake and deposition, suggesting that only ventilation rate V_E was varied by mode in the uptake model.

Bicyclists' uptake of traffic-related VOC was directly measured by sampling blood and urine concentrations of BTEX compounds (benzene, toluene, ethylbenzene, and xylenes) by Bergamaschi et al. (1999). They found significant increases of benzene and toluene in blood for bicyclists in urban areas, and significant increases of toluene and xylenes in urine. Although uptake was directly measured, respiration was not measured, and there was no discussion of pollutant intake or inhalation, which inhibits placement of their findings in the larger context of the emissions-health pathway (Figure 1). Nwokoro et al. (2012) directly measured uptake doses of BC by bicyclists and non-bicyclists (pedestrians and public transit riders) in London by sampling airway macrophages. They found significantly higher (63%) doses of BC for bicyclists, correlated with higher commute exposure concentrations. Bicyclists also had almost twice as long commute durations, and experienced 41% of daily BC exposure during the commute (as compared to 19% for non-bicyclists).

The few studies of bicyclists' pollution uptake suggest that PM uptake doses are disproportionally greater for bicyclists than intake doses or exposure concentrations when compared to other modes. Bicyclists' uptake doses of gaseous pollutants are also disproportionately higher than exposure concentrations when compared to other modes, but have yet to be directly compared to intake doses. Uptake dose is the closest measure of health risks for exposed travelers, but connections to health outcomes still require application of a dose-response function that reflects the toxicity of the pollutants, the susceptibility of the travelers and other factors (Cho et al., 2009; ICRP, 1994).

6 Health Effects of Bicyclists' Air Pollution Uptake

Linkages between long-term exposure to traffic-related air pollution and health impacts have been established, as described elsewhere (Bell, 2012; Brook et al., 2010; Brugge, Durant, & Rioux, 2007; Health Effects Institute, 2010; Nawrot et al., 2011; Pope & Dockery, 2006; Samet, 2007). Long-term health effects studies show elevated risk for development of asthma, reduced lung function, increased blood pressure, and cardiac and pulmonary mortality. An important gap for traveler health studies, though, is a lack of data on the health effects of chronic high-intensity but short-duration doses (Zuurbier, Hoek, Oldenwening, Meliefste, Krop, et al., 2011). Some evidence exists of effects on mortality and cardiovascular/pulmonary hospital admissions for short-term exposure to traffic-related air pollution in general, and particularly PM and UFP (Knibbs et al., 2011; McCreanor et al., 2007; Michaels & Kleinman, 2000; Peters et al., 2004). A recent study indicates increased risk of acute myocardial infarction onset after travel specifically for bicyclists – though the risk is not higher than for other modes (Peters et al., 2013).

Health effects studies of bicyclists' exposure to air pollution have focused on respiratory and cardiovascular biomarkers following acute (0.5-2 hour) exposures to traffic (11 studies of Types K-M in Table 3). Biomarkers are physiological indicators in the pathway of the morbidity and mortality outcomes studied in epidemiology; for example, blood cell counts can be indicators of systemic inflammation, and systemic inflammation is linked to cardiovascular disease (Brook et al., 2010). Unfortunately, even when acute health effects are recognized in the form of biomarkers, the broader health significance is often not known – especially in the context of chronic daily exposures.

Studies of bicyclists' biomarkers show inconsistent results, with 4 of 11 reporting insignificant acute effects and others reporting s*ome* cardiovascular or respiratory biomarker changes. No significant changes in bicyclists' respiratory or cardiovascular biomarkers were reported in four studies of acute on-road exposure (Jarjour et al., 2013; Waldman et al., 1977; Zuurbier, Hoek, Oldenwening, Meliefste, Krop, et al., 2011; Zuurbier, Hoek, Oldenwening, Meliefste, van den Hazel, et al., 2011). Jacobs et al. (2010) found a significant but small increase in a single indicator of blood inflammation for bicyclists, with "unclear" health implications. Cole-Hunter et al. (2013) found significant differences in nasal and throat irritation between bicyclists in high-exposure and low-exposure routes, but no significant differences for airway inflammation biomarkers. Strak et al. (2010) found mostly insignificant changes in respiratory function biomarkers for bicyclists, though UFP and soot exposure were weakly associated with a biomarker of airway inflammation (exhaled NO) and degraded lung function. Weichenthal et al. (2011) found significant associations between UFP, ozone (O_3), and NO_2 exposures during travel and cardiovascular risk indicators (changes in heart rate variability), but no strong associations between in-traffic exposure and respiratory biomarkers. Further analysis of individual VOC in the data set found "evidence of possible associations … for a small number of compounds" with biomarkers of lung inflammation, lung function, and heart rate variability (Weichenthal et al., 2012). Nyhan et al. (2014) found significant associations between decreased heart rate variability and $PM_{2.5}$ and PM_{10} doses – stronger for bicyclists and pedestrians than other modes. Bos et al. (2011) took a different approach and found that PM exposure during bicycling can suppress a *positive* exercise-induced health biomarker associated with cognitive performance. Though again, the effects of chronic exposure are still unknown.

This review does not address the health impacts of bicycling-related crashes and physical activity, only air pollution uptake. However, a review of five recent health impact assessments for bicycling concludes that the physical activity benefits of bicycling far outweigh the crash safety and air pollution risks – by factors of 9 to 96 (Teschke, Reynolds, Ries, Gouge, & Winters, 2012). The air pollution risks in these assessments are based on extrapolations of epidemiological evidence for long-term health outcomes, and limited by the continued uncertainty of health effects of chronic daily uptake of air pollution by physically active travelers.

7 Summary

This is the first review to specifically address bicyclists' health risks from traffic-related air pollution and to explicitly include intake and uptake doses in addition to exposure concentrations. Bicyclists' pollution **exposure concentrations** are highly variable, with median increases of up to 102% (for gaseous hydrocarbons) on high traffic versus low traffic routes. Bicyclists' relative exposure concentrations compared to other modes are inconsistent, varying by pollutant, facility, route, and city. Bicyclists' exposure concentrations are most affected by wind and proximity to motor vehicle traffic, though few studies have incorporated detailed, concurrent traffic data.

Bicyclists' pollution **intake doses** tend to be higher than motorized modes due to their 2 to 5 times higher respiration rates. Bicyclists' respiration and intake dose increase with bicycle travel speed and exertion, but only 12 of the 57 studies with spatially-explicit bicyclist exposure concentration data include any measurement of respiration. Furthermore, only 3 of those studies consider variable bicyclist respiration rates, and there has been almost no assessment of the variability in respiration with trip characteristics (including correlation with exposure concentrations).

Bicyclists' pollution **uptake doses** are affected by the intake dose, pollutant characteristics, breathing depth and pathway, and other individual and physiological factors. Uptake rates tend to increase with exertion level, affecting bicyclists more than motorized travelers. There are clear links between traffic-related air pollution exposure and negative **health** outcomes in urban populations. However, the health effects of chronic daily air pollution uptake by bicyclists are still unknown. More research is needed on health impacts of pollution exposure because some studies of bicyclists' biomarkers show significant acute respiratory effects while other studies show insignificant effects.

To reduce exposure concentrations, spatial and temporal separation of bicyclists from motor vehicle traffic can be achieved with separated bicycle facilities, low-volume routes, and off-peak travel. These are potential "win-win" strategies because bicyclists already prefer low-traffic routes and bicycle-specific facilities (Broach et al., 2012; Dill, 2009; Kang & Fricker, 2013; Wardman, Tight, & Page, 2007) and separated bicycle facilities could also improve safety (Lusk et al., 2011; Reynolds, Harris, Teschke, Cripton, & Winters, 2009; Teschke, Harris, et al., 2012). Regarding intake doses, other likely mitigation strategies would be to prioritize separation from traffic in locations where bicyclists' respiration is expected to be high (steep grades, for example) or to reduce energy expenditure requirements (by reducing required stops, for example) in locations where pollutant concentrations are known to be high.

8 Research Gaps and Opportunities

This literature review reveals steady progress towards a better understanding of air pollution uptake by bicyclists. However, several significant research gaps deserve attention. Although the literature suggests that traffic-related air pollution uptake is higher for bicyclists than for travelers using motorized modes, persistent uncertainty in the intensity and effects of pollution uptake means that transportation planners and decision makers are unable to consider bicyclists' air pollution risks in a precise way. More research is needed to provide better quantification and understanding of the relative health benefits of alternative bicycle facility designs, bicycle network designs, and route options. Some research topics that can bring us closer to achieving these goals include:

- Study of the on-road variability of respiration and air quality for traveling bicyclists, including a broader array of pollutants (e.g. ground-level ozone);
- The impact of bicycle trip attributes such as road grade, road surface, travel speed, and number of stops on respiration rates for bicyclists;
- The impacts of bicycle facility design features on exposure concentrations (distance from motor vehicle travel lanes, physical barriers, intersection treatments such as "bike boxes", etc.);
- The impacts of traffic flow characteristics on bicyclists' exposure concentrations, including traffic speeds, volumes, and queuing along arterials or at major intersections;

- Inter-modal pollution exposure comparisons that apply more comprehensive and representative modal travel characteristics (trip location and distance, traveler demographics, route preferences) and that consider variable respiration (especially for active travelers);
- Characterization of different bicyclist types (e.g. commuters, recreational riders) and demographic factors that can impact respiration or health effects; these factors include physiology (height, weight, respiratory health), riding style (speed, acceleration, response to grades), and equipment (weight, condition, baggage);
- Analysis of bicyclists' pollutant doses along different types of routes and facilities, to enable health impact assessments; and
- Development of dose-response functions for health effects of chronic short-duration high-intensity air pollution exposure episodes.

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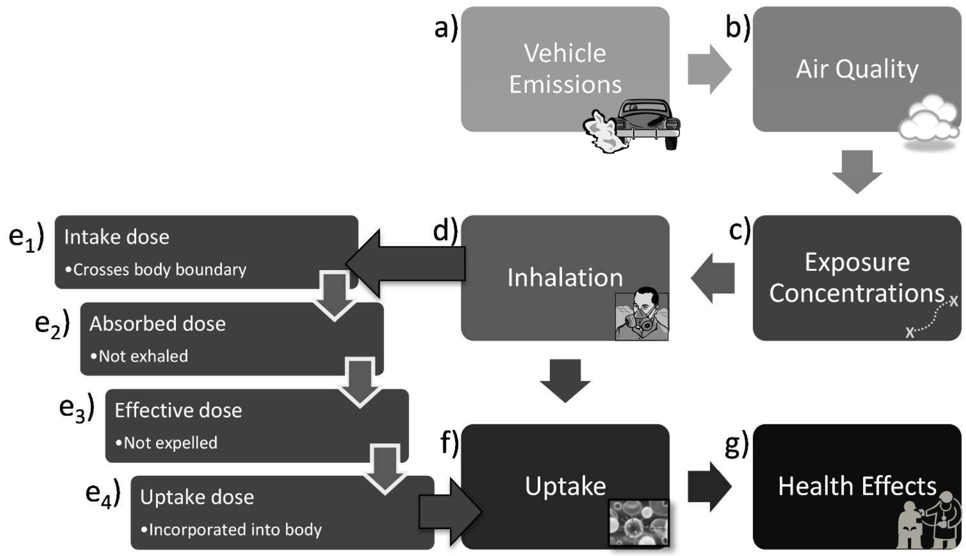

Figure 1. Conceptual Diagram of Exposure Pathway for Traffic-Related Air Pollution; adapted from Ott, Stieneman, & Wallace (2007)
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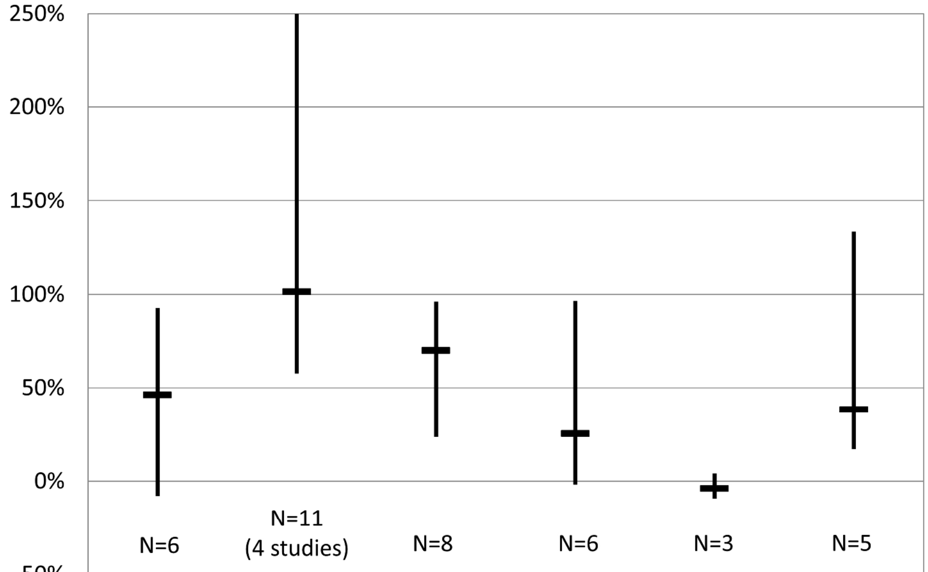

	CO	VOC	UFP	PM2.5	PM10	BC
High	92%	250%	96%	97%	4%	134%
Low	-8%	58%	24%	-2%	-9%	17%
‒ Median	47%	102%	70%	26%	-4%	39%

254x190mm (300 x 300 DPI)

Appendix – Supplemental Material

Table of Contents – Supplemental Material

1 LITERATURE SEARCH METHOD

1.1 Database Search

A systematic literature search for bicyclist exposure and dose measurements was performed through January 2014 using all 20 possible combinations $\{A + B + C\}$ of the keywords $A = \{\text{bicycle}, \text{bicyclist}, \text{cyclist}, \text{bike}\}$, $B = \{\text{pollution}\}$ and $C = \{\text{exposure}, \text{intake}, \text{inhalation}, \text{uptake}, \text{dose}\}$. An exhaustive search was performed of the WorldCat™ catalogue. The number of hits returned for each search phrase ranged from 0 ("bicyclist pollution intake") to 131 ("bicycle pollution exposure"); 231 unique hits were returned. The same 20 search phrases were used with the Google Scholar™ search engine. Because of the volume of Google Scholar™ hits returned (28,100 for "bicycle pollution exposure" alone), only the first 50 hits per search phrase were processed (sorted by relevance).

1.2 Filtering

Of the 231 unique hits returned from the WorldCat™ database search, a first screening was performed with exclusion based on title review or reference format (theses, conference papers, and textbooks were excluded). This screening removed 119 hits, leaving 112 potential papers. A matching exercise was then performed to remove further duplicate papers – resulting in 47 duplicates removed. Another 11 papers were excluded based on abstract review, leaving 54 papers for full-text extraction. The title and abstract review process required that papers describe original studies about on-road bicyclists and environmental air pollution exposures. Reviews, chamber studies using bicycle ergometers, and traveler exposure studies not including bicyclists were excluded. The citation lists of these 54 papers and the Google Scholar™ search returns were searched for additional papers that passed the same format, title review, and abstract review criteria. The result was 14 additional papers manually added to the full-text body of references, now composed of 68 papers.

1.3 Classification of Papers

The full-text body of 68 references was reviewed for two nested inclusion criteria. The first criterion was the use of spatially-explicit concentration data, either measured or modeled. Studies that assumed a generic concentration value (de Hartog et al., 2010) were excluded. 57 papers met this criterion. The second criterion was the presentation of original exposure concentration data, measured on-road by bicyclists. Studies using modeled concentration data, roadside monitor data, conducting analysis using previously-published exposure concentration data, or not reporting central value statistics were excluded. 42 papers met this criterion. If multiple papers reported on the same data set, a single reference was included in this subset. Two studies measured bicyclists' exposures but were focused on instrument development and did not report central value statistics (Elen et al., 2013; Piechocki-Minguy et al., 2006). The literature search method is summarized in Figure 1.

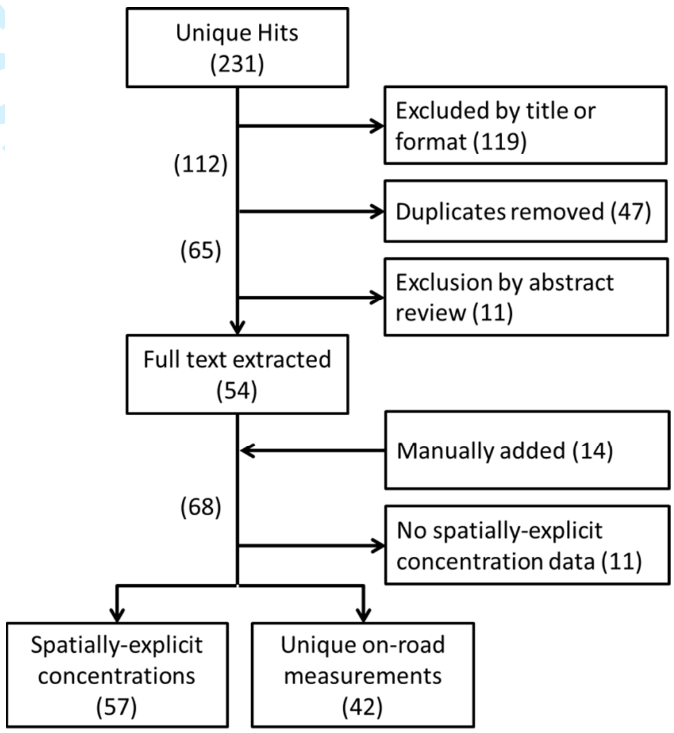

Figure 1. Literature Search Summary

2 TRAFFIC-RELATED AIR POLLUTANTS

This section briefly introduces the main traffic-related air pollutants linked to health risks for road travelers. Most primary traffic-related air pollutants are combustion by-products emitted from the tailpipes of motor vehicles; other sources include evaporation, brake and tire wear, and resuspension of road dust. As vehicle engine exhaust becomes cleaner, brake and tire wear may be a growing portion of vehicle-related urban particulate matter. Secondary traffic-related air pollutants are formed through atmospheric physical and chemical processes, after the emission of primary pollutants. Detailed information on these pollutants is readily found in textbooks on air pollution such as Vallero (2007). Several important traffic-related air pollutants are excluded from this review because they have not been directly measured on-road in bicyclist exposure studies, including ground-level ozone (O_3) and sulfur oxides (SO_x). This is particularly troublesome for O_3, a secondary pollutant associated with numerous health effects including neurodegeneration, respiratory and cardiovascular morbidity, and mortality (Health Effects Institute, 2010; U.S. Environmental Protection Agency, 2013a).

Carbon monoxide (CO) is emitted by vehicles as a result of incomplete combustion of hydrocarbon fuel. Transportation microenvironments tend to have elevated concentrations of CO (El-Fadel and Abi-Esber, 2009; Kaur et al., 2007). Even at ambient levels CO has known negative health effects such as exacerbation of heart disease and neurological damage, with little to no evidence of safe threshold concentrations (Burnett et al., 1998; Ott et al., 2007; Townsend and Maynard, 2002; U.S. Environmental Protection Agency, 2010; World Health Organization, 1999).

Nitrogen oxides (NO_x) are another major component (by mass) of motor vehicle primary pollution emissions. NO_x is emitted in the forms of nitric oxide (NO) and nitrogen dioxide (NO_2); NO_x composition changes through secondary reactions with ozone and other oxidants (Carslaw and Beevers, 2005; Tian et al., 2011). Short-term NO_2 exposure, even at ambient levels, has been associated with adverse respiratory effects and mortality rates (Samoli et al., 2006; U.S. Environmental Protection Agency, 2008).

Volatile organic compounds (VOC) are commonly emitted through vehicle exhaust, engine evaporation, and refueling evaporation (Brown et al., 2007; Gertler et al., 1996). VOC is a broad category including many individual gas compounds such as hydrocarbons in fuel (octane, benzene), fuel additives (ethers such as MTBE), and combustion byproducts (acrolein, formaldehyde). Motor vehicles are a major source of gaseous hydrocarbons and other VOC in urban areas (Brown et al., 2007; Kansal, 2009; Watson et al., 2001). The U.S. Federal Highway Administration (FHWA) and Environmental Protection Agency (EPA) have identified seven high-priority mobile source air toxics with “significant contributions” to cancer risk; six of these air toxics are VOC: acrolein, benzene, 1,3-butidiene, formaldehyde, naphthalene, and other polycyclic organic matter (U.S. Federal Highway Administration, 2009).

Particulate Matter (PM) air pollution includes particles of varying size and composition, often composed of dissimilar molecules. Disproportionately large fractions of total daily exposure to PM occur during commuting (Dons et al., 2012; Fruin et al., 2008; Nwokoro et al., 2012; Ragettli et al., 2013). Particulate matter is often categorized by its size. The smallest size category commonly studied in the exposure literature is ultrafine particles (UFP): particles with aerodynamic diameters below 100 nm. Larger particulate matter is designated PM_x, where x is the maximum aerodynamic diameter. Two important size categories are $PM_{2.5}$ (“fine”) and PM_{10} (“inhalable”) – both of which are subject to ambient air quality standards due to their known negative health effects (Ott et al., 2007; U.S. Environmental Protection Agency, 2012). PM_x categories are reported as mass concentrations but UFP are typically reported as particle number concentrations (PNC). In contrast to the size categories, elemental carbon (EC) and black carbon (BC) are terms for soot particles generated as a combustion byproduct (Andreae and Gelencsér, 2006).

The larger PM size categories have better-established monitoring data and more robust epidemiological evidence for health outcomes such as cardiopulmonary morbidity and mortality (Brook et al., 2010). $PM_{2.5}$ is thought to have larger impacts on health than PM_{10} because of more toxic composition and deeper penetration in the lungs; $PM_{2.5}$ appears to have no safe concentration threshold for exposure (Pope and Dockery, 2006). Similarly, UFP have received increasing attention as a health risk because of their size (allowing deep lung penetration and entry to the bloodstream) and composition (high surface area and reactive compounds) (Knibbs et al., 2011). The larger particles have more biogenic sources and a smaller proportion of ambient concentrations are due to primary emissions from motor vehicles than for smaller PM. High UFP number concentrations are often found in transportation microenvironments (Knibbs et al., 2011), and the UFP size category dominates total PNC in near-road environments

(Morawska et al., 2008). However, UFP emissions models, monitoring data, and epidemiological evidence are all still lacking when compared to larger PM size categories.

3 SUMMARY OF MEASURED BICYCLIST EXPOSURE CONCENTRATIONS

The following Tables (S.1 through S.8) summarize all 42 on-road bicycle exposure monitoring studies, grouped by pollutant. In cases where the same original data set appears in more than one publication, a single citation is included in the tables.

The earliest studies measured CO exposure concentrations for bicyclists in U.S. cities (Kleiner and Spengler, 1976; Waldman et al., 1977). The first multi-pollutant study measured CO, $PM_{3.5}$, and VOC for bicyclists in the UK (Bevan et al., 1991). Since then the majority of on-bicycle pollution exposure studies have taken place in Europe, North America, Australia, and New Zealand, with a few recent exceptions from China (Huang et al., 2012; Yu et al., 2012). In addition to the on-bicycle data collections included in these tables, several other studies approximated bicyclists' exposure concentrations using stationary near-road (or near-path) measurements of PM_{10} (Fajardo and Rojas, 2012), UFP (Kendrick et al., 2011), NO_2 (Bean et al., 2011), and CO and NO_x (Chan et al., 1994).

To provide context for the values in these tables, World Health Organization guidelines for annual mean $PM_{2.5}$, PM_{10}, and NO_2 concentrations are 10, 20, and 40 μg/m^3, respectively (Krzyzanowski and Cohen, 2008). A review of UFP measurements suggests an "urban background" concentration of 7,290 pt/cc (Morawska et al., 2008). Ambient monitoring in U.S. cities shows typical annual 2nd maximum 8-hour average ambient CO concentrations of 1.5 ppm in 2012 (U.S. Environmental Protection Agency, 2013b); annual average CO concentrations would be much lower (Wang et al., 2011). Pankow et al. (2003) measured ambient VOC levels in U.S. urban areas as 0.12-1.1 μg/m^3 for benzene, 0.39-2.7 μg/m^3 for toluene, and 0.54-1.6 μg/m^3 for xylenes. Although a multi-city background study for BC was not found, deCastro et al. (2008) estimate a representative urban background BC concentration of 0.9 μg/m^3 for a U.S. city, while monitoring in a Belgian city measured median background BC concentrations around 1.5 μg/m^3 (Dekoninck et al., 2013).

Table S.1 On-road Measured Bicyclist Exposure Concentrations of CO (ppm)

Reference	Location (setting)	Method[1]: Instruments	Modes (and/or bicycle routes) compared	Time frame	# on-bicycle samples	Central value [2,3]	Variability [2,4]
(Bevan et al., 1991)	Southampton, UK (urban, suburban)	EC: sample pump with Neotronics sensor	None (urban and suburban routes)	Unspecified dates; weekdays (assumed); 800-900 and 1630-1730	16 trips	10.5	5.3-17.9
(Dirks et al., 2012)	Aukland, New Zealand (urban)	EC: Langan T15n	Run, bus, train, auto	Unspecified dates; weekdays (assumed); AM and PM peak periods	4 trips	0.6	NR
(Hatzopoulou et al., 2013)	Montreal, Canada (urban)	EC: Langan T15n	None	May-August 2011; weekdays (Mon-Thurs); AM: 800-1000 and PM: 1500-1700	61 trips (including AM and PM trips)	0.96 (AM); 1.22 (PM)	0.56-1.3 (AM); 0.83-1.75 (PM)
(Huang et al., 2012)	Beijing, China (urban)	EC: Langan T15n	Bus, taxi	December 2010-February 2011; weekdays; peak and off-peak periods	43 trips	1.90	1.00-3.39
(Jarjour et al., 2013)	Berkeley, USA (urban, suburban)	NDIR: TSI Q-Trak 7565	None (low-traffic and high-traffic routes)	April-June 2011; weekdays; 800-1000	8 low-traffic, 10 high-traffic trips	0.79 (low-traffic); 0.90 (high-traffic)	0.20-4.90 (low-traffic); 0.10-10.60 (high-traffic)
(Kaur et al., 2005)	London, UK (urban)	EC: Langan T15 and T15v	Walk, bus, auto, taxi	April-May 2003; unspecified days; 830-315 (3-4x/day)	29 trips	1.1	0.2-2.9
(Kingham et al., 2013) [5]	Christchurch, New Zealand (urban)	EC: Langan T15n	Bus, auto (on-road and off-road routes)	February-March 2009; weekdays; 740-900 and 1645-1805	49 on-road, 48 off-road trips	Median: 0.7 (on-road); 0.5 (off-road)	0.1-2.9 (on-road); 0.1-2.1 (off-road)
(Kleiner and Spengler, 1976)	Boston, USA (urban)	EC: Ecolyzer series 2000	None (peak/off-peak and road type comparisons)	Summer 1976; unspecified days; peak and off-peak periods	176 trips	12.6	SD: 4.8
(de Nazelle et al., 2012)	Barcelona, Spain (urban)	NDIR: TSI Q-Trak 7565	Walk, bus, auto	May-June 2009; weekdays; 800-2000 (peak and off-peak)	38 trips	GM: 1.5	GSD: 1.6
(Waldman et al., 1977)	Washington D.C., USA (urban)	NDIR: EMI sample pump; Tedlar bags to 1.5 L; Beckman 865 NDIR	Auto (4 routes: high/low traffic and high/low density)	May-July 1977; weekdays; PM peak period	52 trips	8.2	0.9-21.0
(Weichenthal et al., 2011)	Ottawa, Canada (urban)	EC: Langan T15n	None (low-traffic and high-traffic routes, indoors)	May-September 2010; weekdays; 1130-1230	≤40 low-traffic, ≤39 high-traffic trips	0.9 (low-traffic); 1. 4 (high-traffic)	0.5-1.5 (low-traffic); 0.6-2.6 (high-traffic)
(van Wijnen et al., 1995)	Amsterdam, Netherlands (urban, rural)	GC: sample pump (0.1 L/min); 10-L Tedlar bags; GC with FID	Walk, auto (urban and rural routes)	January, May, and August 1990; weekdays; 800-1000 and 1500-1800	16 rural, 66 urban trips	<0.5 (rural); 1.6 (urban)	all <0.5 (rural); <0.5-3.6 (urban)

1. Method abbreviations: EC=electrochemical, NDIR=non-dispersive infrared, GC=gas chromatography
2. Concentration units are ppm; values reported as μg/m^3 (van Wijnen et al., 1995) are converted to ppm using a molar gas volume of 24.45 L
3. Central values are arithmetic means unless otherwise noted (median, GM=geometric mean)
4. Variability is expressed as the range unless otherwise noted (SD=standard deviation, GSD=geometric standard deviation, NR=not reported)
5. Additional data for Kingham et al. (2013) retrieved from Kingham et al. (2011)

Table S.2. On-road Measured Bicyclist Exposure Concentrations of Individual VOC ($\mu g/m^3$)

Reference	Location (setting)	Instruments	Modes (and/or bicycle routes) compared	Time frame	# on-bicycle samples	Central value [1,2,3]	Variability [1,2,4]
(Bergamaschi et al., 1999)	Italy (unspecified location; urban/rural)	Radiello passive diffusive sampler	None (urban and rural routes)	December 1997-February 1998; unspecified days; unspecified hours	24 rural, 24 urban trips	Benzene (median): 6.2 (rural); 48.2 (urban) Toluene (median): 13.5 (rural); 113.0 (urban) Ethylbenzene (median): 3.8 (rural); 19.8 (urban) Xylenes (median): 21.8 (rural); 105.2 (urban)	 3.8-12.3 (rural); 22.5-83.6 (urban) 6.7-64.2 (rural); 45.0-1230.0 (urban) 1.3-10.2 (rural); 10.2-30.6 (urban) 5.4-61.5 (rural); 12.5-183.0 (urban)
(Bevan et al., 1991)	Southampton, UK (urban, suburban)	Diaphragm pump (0.1 L/min); ATD-50 sorption tube with Tenax TA; ATD and GC/MS	None (urban and suburban routes)	Unspecified dates; weekdays (assumed); 800-900 and 1630-1730	16 trips	Benzene: 56 Ethylbenzene: 23 Toluene: 122 m,p-xylenes: 64 o-xylenes: 33	19-120 8-58 56-279 24-115 9-166
(Chertok et al., 2004)	Sydney, Australia (urban)	Custom BTEX passive sampler tube	Walk, bus, train, auto	September 2002; weekdays; peak periods	14 samples, 10 commute trips per sample	Benzene (GM): 19.71 Toluene (GM): 92.55 Ethylbenzene (GM): 11.81 Xylenes (GM): 52.80	NR NR NR NR
(Kingham et al., 1998)	Huddersfield, UK (small city)	Sample pump; ATD sorption tube	Bus, train, auto (on-road and cycle path routes)	September-October 1996; Mon-Sat; unspecified times	6 sample days	Benzene: 15.7 (cycle path); 26.5 (on-road)	 4.8-50.7 (cycle path); 5.5-74.6 (on-road)
(McNabola et al., 2008)	Dublin, Ireland (urban)	SKC vacuum pump; 1-L Tedlar bag; ATD and GC/MS with FID	Walk, bus, auto (2 routes and urban/ suburban comparison)	January 2005-June 2006; weekdays; 800-900 and 1700-1800	42 trips route 1, 43 trips route 2	Benzene: 5.49 (route 1); 4.92 (route 2)	SD: 2.30 (route 1); 2.59 (route 2)
(O'Donoghue et al., 2007)	Dublin, Ireland (urban)	SKC vacuum pump; 1-L Tedlar bag; ATD and GC/MS with FID	Bus	February 6-14 2003; weekdays (implied); 800-1000 and 1600-1800	14 samples of ~13 min	Benzene: 5.18	1.73-9.14
(Rank et al., 2001)	Copenhagen, Denmark (urban)	Sampling pump (1.9 L/min); charcoal sorption tube; GC/MS	Auto	18 June and 3 August 1998; weekdays (Thurs and Mon); 740-940 and 1000-1200	4 samples of ~2 hr	Benzene: 5.2 Toluene: 20.6 Ethylbenzene, xylenes: 18.1	4.5-5.6 19.4-22.9 9.9-23.3

(Weichenthal et al., 2011) [6]	Ottawa, Canada (urban)	SUMMA canisters (1 L); GC/MS	None (low-traffic and high-traffic routes, indoors)	May-September 2010; weekdays; 1130-1230	40 low-traffic, 39 high-traffic trips	Benzene (median): 0.94 (low-traffic); 0.34 (high-traffic) Toluene (median): 1.1 (low-traffic); 3.4 (high-traffic) M,p-xylenes (median): 0.4 (low-traffic); 1.4 (high-traffic) o-xylenes (median): 0.2 (low-traffic); 0.5 (high-traffic)	 0.2-3.5 (low-traffic); 0.1-0.9 (high-traffic) 0.3-83 (low-traffic); 0.4-22 (high-traffic) 0.1-3.4 (low-traffic); 0.2-43 (high-traffic) 0.05-0.8 (low-traffic); 0.08-6.9 (high-traffic)
(van Wijnen et al., 1995) [5]	Amsterdam, Netherlands (urban, rural)	Dupont continuous flow pump (1 L/min); SKC active carbon sorption tube; GC/MS with FID	Walk, auto (urban and rural routes)	January, May, and August 1990; Weekdays; 800-1000 and 1500-1800	16 rural, 74 urban trips	Benzene: <8 (rural); 19 (urban) Toluene: 13 (rural); 51 (urban) Xylenes : <8 (rural); 36 (urban)	all <8 (rural); <8-44 (urban) <8-23 (rural); 19-106 (urban) all <8 (rural); 15-75 (urban)

1. Only reported BTEX values are included; other compounds are reported in some studies (Bevan et al., 1991; McNabola et al., 2008; O'Donoghue et al., 2007; Weichenthal et al., 2011)
2. Concentration units are $\mu g/m^3$; values reported as ppb (Chertok et al., 2004; McNabola et al., 2008; O'Donoghue et al., 2007) are converted using a molar gas volume of 24.45 L
3. Central values are arithmetic means unless otherwise noted (median, GM=geometric mean)
4. Variability is expressed as the range unless otherwise noted (SD=standard deviation, NR=not reported)
5. van Wijnen et al. (1995) report separate values for each month/route combination; the values in the table are averages weighted by number of samples
6. Additional data for Weichenthal et al. (2011) retrieved from Weichenthal et al. (2012)

Table S.3. On-road Measured Bicyclist Exposure Concentrations of NO_2 ($\mu g/m^3$)

Reference	Location (setting)	Instruments	Modes (and/or bicycle routes) compared	Time frame	# on-bicycle samples	Central value [1,2]	Variability [1,3]
(Bernmark et al., 2006)	Stockholm, Sweden (urban bicycle couriers)	Passive sampler	None	Unspecified dates; weekdays; working hours	5 samples of 8-hr work-days	51	40-60
(Chertok et al., 2004)	Sydney, Australia (urban)	Custom passive sampler	Walk, bus, train, auto	September 2002; weekdays; peak periods	14 samples, 10 commute trips per sample	GM: 46.25	NR
(Farrar et al., 2001)	Perth, Australia (urban)	Custom passive sampler	Bus, taxi (commuters and couriers)	Unspecified dates; weekdays (assumed); peak periods for commuters, work days for couriers	8 commuter, 15 courier samples of 24 hours	41 (commuter); 26 (courier)	17-60 (commuter); 8-70 (courier)
(van Wijnen et al., 1995)	Amsterdam, Netherlands (urban/rural)	Dupont continuous flow pump (0.2 L/min); SKC sample tube for NO_2; desorption and spectrophotometry	Walk, auto (urban and rural routes)	January, May, and August 1990; Weekdays; 800-1000 and 1500-1800	4 rural, 27 urban trips	90 (rural); 114 (urban)	<60-267 (rural); <60-262 (urban)

1. Concentration units are $\mu g/m^3$; values reported as ppb (Chertok et al., 2004; Farrar et al., 2001) are converted using a molar gas volume of 24.45 L
2. Central values are arithmetic means unless otherwise noted (GM=geometric mean)
3. Variability is expressed as the range unless otherwise noted (NR=not reported)

Table S.4. On-road Measured Bicyclist Exposure Concentrations of UFP and PNC [1] (pt/cc)

Reference	Location (setting)	Instruments	Modes (and/or bicycle routes) compared	Time frame	# on-bicycle samples	Central value [2]	Variability [3]
(Berghmans et al., 2009)	Mol, Belgium (small city)	TSI P-Trak 8525	None	April 2007; weekdays (Mon-Thurs); 0600-1600	358 1-min samples	21,226	5,429-122,000
(Boogaard et al., 2009)	11 medium-size Dutch cities	TSI CPC 3007	Auto	August-October 2006; weekdays (Mon-Thurs); 1200-1900	1,536 1-min samples	24,329	5,103-112,219
(Cole-Hunter et al., 2012)	Brisbane, Australia (urban)	Philips Aerasense NanoTracer	None (low-traffic and high-traffic routes)	December 2010-January 2011; weekdays; 700-800	12 low-traffic, 12 high-traffic trips	15,600 (low-traffic); 30,600 (high-traffic)	SD: 3,800 (low-traffic); 5,300 (high-traffic)
(Cole-Hunter et al., 2013)	Brisbane, Australia (urban)	Philips Aerasense NanoTracer	None (low-traffic and high-traffic routes)	Unspecified dates; weekdays; peak periods	70 low-traffic, 70 high-traffic trips	19,100 (low-traffic); 29,500 (high-traffic)	SD: 9,300 (low-traffic); 15,000 (high-traffic)
(Hatzopoulou et al., 2013)	Montreal, Canada (urban)	TSI CPC 3007	None	May-August 2011; weekdays (Mon-Thurs); 800-1000 and 1500-1700	51 trips (including AM and PM trips)	24,800 (AM); 21,800 (PM)	13,500-41,000 (AM); 7,600-60,100 (PM)
(Int Panis et al., 2010)	Brussels (B), Louvain-la-Neuve (L), and Mol (M), Belgium (small to large cities)	TSI P-Trak 8525	Auto (routes in 3 cities compared)	June-July 2009; weekdays; unspecified times	24 trips in B, 6 trips in L, 13 trips in M	30,214 (B); 11,865 (L); 8,734 (M)	SD: 9,173 (B); 3,129 (L); 2,496 (M)
(Jacobs et al., 2010)	Antwerp, Belgium (cycle track near a major roadway)	TSI P-Trak 8525	None (clean room)	May 2009; unspecified days; 800-1700	38 trips	28,867	SD: 8,479
(Jarjour et al., 2013)	Berkeley, USA (urban, suburban)	TSI CPC 3007	None (low-traffic and high-traffic routes)	April-June 2011; weekdays; 800-1000	9 low-traffic, 9 high-traffic trips	14,311 (low-traffic); 18,545 (high-traffic)	2,771-376,495 (low-traffic); 1,900-1,033,188 (high-traffic)
(Kaur et al., 2005)	London, UK (urban)	TSI P-Trak 8525	Walk, bus, auto, taxi	April-May 2003; unspecified days; 830-315 (3-4x/day)	21 trips	93,968	53,127-178,601
(Kingham et al., 2013) [4]	Christchurch, New Zealand (urban)	TSI CPC 3007	Bus, auto (on-road and off-road routes)	February-March 2009; weekdays; 740-900 and 1645-1805	44 on-road, 34 off-road trips	Median: 31,414 (on-road); 16,641 (off-road)	10,121-160,520 (on-road); 3,601-81,626 (off-road)
(de Nazelle et al., 2012)	Barcelona, Spain (urban)	TSI CPC 3007	Walk, bus, auto	May-June 2009; weekdays; 800-2000 (peak and off-peak)	46 trips	GM: 75,300	GSD: 1.2
(Quiros et al., 2013)	Santa Monica, USA (urban residential area near ocean)	TSI CPC 3007	Walk, auto	March-April 2011; weekdays and weekends; 730-930, 1230-1430, and 1700-1900	27 samples of ~2 hr (3x per day for 9 days)	GM: 31,800 (morning), 10,600 (afternoon), 13,200 (evening)	GSD: 1.85 (morning), 1.77 (afternoon), 1.98 (evening)
(Ragettli et al., 2013)	Basel, Switzerland (urban)	miniature Diffusion Size Classifier (miniDiSC)	Walk, bus, tram, auto[5] (low-traffic and high-traffic routes)	Winter, Spring, and Summer 2011; weekdays; AM and PM peak periods	36 low-traffic, 36 high-traffic trips	18,156 (low-traffic); 34,025 (high-traffic)	SD: 8,615 (low-traffic); 26,406 (high-traffic)
(Strak et al., 2010)	Utrecht, Netherlands (urban)	TSI CPC 3007	None (low-traffic and high-traffic routes)	April-May 2007; weekdays (Mon-Thurs); 800-930	12 low-traffic, 12 high-traffic trips	27,813 (low-traffic); 44,090 (high-traffic)	18,047-38,796 (low-traffic); 28,443-58,409 (high-traffic)

(Thai et al., 2008)	Vancouver, Canada (urban)	TSI P-Trak 8525	None	August-October 2007; weekdays; 700-900	7 sample days	33,899	18,830-57,692
(Vinzents et al., 2005)	Copenhagen, Denmark (urban)	TSI CPC 3007	None (indoors and other outdoor activities)	March-June 2003; weekdays; AM and PM peak periods	74 samples of ~1.5 hr	GM: 32,400	GSD: 1.49
(Weichenthal et al., 2011)	Ottawa, Canada (urban)	TSI CPC 3007	None (low-traffic and high-traffic routes, indoors)	May-September 2010; weekdays; 1130-1230	40 low-traffic, 39 high-traffic trips	10,882 (low-traffic); 19,747 (high-traffic)	3,590-34,000 (low-traffic); 6,834-27,800 (high-traffic)
(Zuurbier et al., 2010)	Arnhem, Netherlands (urban)	TSI CPC 3007	Bus, auto (low-traffic and high-traffic routes)	June 2007 – June 2008; weekdays (Tues and Thurs); 800-1000	15 low-traffic, 15 high-traffic trips	39,576 (low-traffic); 48,939 (high-traffic)	SD:18,178 (low-traffic); 19,039 (high-traffic)

1. Studies reporting PNC and UFP are combined because of the dominance of the UFP size range in near-road PNC (Morawska et al., 2008)
2. Central values are arithmetic means unless otherwise noted (Median, GM=geometric mean)
3. Variability is expressed as the range unless otherwise noted (SD=arithmetic standard deviation, GSD=geometric standard deviation)
4. Additional data for Kingham et al. (2013) retrieved from Kingham et al. (2011)
5. Ragettli et al. (2013) modal comparisons are based on separate measurements (N=51) with unreported bicycle concentrations

Table S.5. On-road Measured Bicyclist Exposure Concentrations of $PM_{2.5}$ (ug/m^3)

Reference	Location (setting)	Method[1]: Instruments[2]	Modes (and/or bicycle routes) compared	Time frame	# on-bicycle samples	Central value [3]	Variability [4]
(Adams et al., 2001)	London, UK (urban)	G: custom high-flow personal sampler (HFPS), Casella sample pump (16 L/min), Pall PTFE filter (37 mm, 2 µm pore)	Bus, rail, auto ("side-street" routes compared)	July and August 1999, February 2000; weekdays; peak and off-peak, 4x/day	40 trips (July), 105 trips (August), 56 trips (February)	34.5 (July), 34.2 (August), 23.5 (February)	13.3-68.7(July), 5.2-129.7(August), 6.8-76.2 (February)
(Berghmans et al., 2009)	Mol, Belgium (small city)	P: Grimm 1.108	None	April 2007; weekdays (Mon-Thurs); 0600-1600	358 1-min samples	38.8	8.72-102
(Boogaard et al., 2009)	11 medium-size Dutch cities	P: TSI DustTrak (unspecified model), no local calibration described in text	Auto	August-October 2006; weekdays (Mon-Thurs); 1200-1900	1,632 1-min samples	44.5	0-452
(Hatzopoulou et al., 2013)	Montreal, Canada (urban)	P: TSI DustTrak (unspecified model), calibration applied from Wallace et al. (2011)	None	May-August 2011; weekdays (Mon-Thurs); 800-1000 and 1500-1700	50 trips (including AM and PM trips)	10.4 (AM); 11.1 (PM)	4.3-28.7 (AM); 2.8-38.2 (PM)
(Huang et al., 2012)	Beijing, China (urban)	G: SKC sample pump (4 L/min), Whatman PTFE filter (37 mm, 2 µm pore); P: LD-6S spectrometer (Beijing Green Tech. Digital)	Bus, taxi	December 2010-February 2011; weekdays; peak and off-peak periods	43 trips	49.10	18.96-112.47
(Int Panis et al., 2010)	Brussels (B), Louvain-la-Neuve (L), and Mol (M), Belgium (small to large cities)	P: TSI DustTrak 8534, unspecified local calibration applied	Auto (routes in 3 cities compared)	June-July 2009; weekdays; unspecified times	24 trips in B, 6 trips in L, 13 trips in M	18.9 (B); 22.7 (L); 44.7 (M)	SD: 5.2 (B); 4.1 (L); 10.9 (M)

(Jacobs et al., 2010)	Antwerp, Belgium (cycle track near a major roadway)	P: Grimm 1.108	None (clean room)	May 2009; unspecified days; 800-1700	38 trips	24.2	SD: 8.7
(Jarjour et al., 2013)	Berkeley, USA (urban, suburban)	P: TSI DustTrak 8520, calibration applied from Yanosky et al. (2002)	None (low-traffic and high-traffic routes)	April-June 2011; weekdays; 800-1000	6 low-traffic, 8 high-traffic trips	4.88 (low-traffic); 5.12 (high-traffic)	2.25-20.96 (low-traffic); 2.25-27.40 (high-traffic)
(Kaur et al., 2005)	London, UK (urban)	G: HFPS – see Adams et al. (2001)	Walk, bus, auto, taxi	April-May 2003; unspecified days; 830-315 (3-4x/day)	48 trips	33.5	9.7-77.5
(Kingham et al., 2013) [5]	Christchurch, New Zealand (urban)	P: Grimm 1.101, 1.107, 1.108	Bus, auto (on-road and off-road routes)	February-March 2009; weekdays; 740-900 and 1645-1805	32 on-road, 46 off-road trips	Median: 16.0 (on-road); 16.3 (off-road)	6.4-38.8 (on-road); 4.8-56.4 (off-road)
(McNabola et al., 2008)	Dublin, Ireland (urban)	G: HFPS – see Adams et al. (2001)	Walk, bus, auto (2 routes and urban/suburban comparison)	January 2005-June 2006; weekdays; 800-900 and 1700-1800	56 trips route 1, 48 trips route 2	88.14 (route 1); 71.61 (route 2)	SD: 61.54 (route 1); 46.94 (route 2)
(de Nazelle et al., 2012)	Barcelona, Spain (urban)	P: TSI DustTrak 8520, no local calibration applied	Walk, bus, auto	May-June 2009; weekdays; 800-2000 (peak and off-peak)	41 trips	GM: 29	GSD: 1.7
(Nyhan et al., 2014)	Dublin, Ireland (urban)	P: Met One Aerocet 531, local calibration using gravimetric analysis	Walk, bus, train	Unspecified dates; weekdays; 800-1000	33 trips	37.1	SD: 30.5
(Quiros et al., 2013)	Santa Monica, USA (urban residential area near ocean)	P: TSI DustTrak 8520, calibration applied from Zhang and Zhu (2010)	Walk, auto	March-April 2011; weekdays and weekends; 730-930, 1230-1430, and 1700-1900	27 samples of ~2 hr (3x/day for 9 days)	10.5 (morning), 7.11 (afternoon), 5.24 (evening)	SD: 7.3 (morning), 4.31 (afternoon), 4.00 (evening)
(Thai et al., 2008)	Vancouver, Canada (urban)	P: Grimm 1.108	None	August-October 2007; weekdays; 700-900	7 sample days	22.6	7.3-33.6
(Weichenthal et al., 2011)	Ottawa, Canada (urban)	P: TSI DustTrak (unspecified model), calibration applied from Wallace et al. (2011)	None (low-traffic and high-traffic routes, indoors)	May-September 2010; weekdays; 1130-1230	~39 low-traffic, ~38 high-traffic trips (unspecified)	8.14 (low-traffic); 12.2 (high-traffic)	2.2-26 (low-traffic); 3.0-34 (high-traffic)
(Zuurbier et al., 2010)	Arnhem, Netherlands (urban)	P: MIE DataRam 1200 with $PM_{2.5}$ cyclone and pump (4 L/min)	Bus, auto (low-traffic and high-traffic routes)	June 2007 – June 2008; weekdays (Tues and Thurs); 800-1000	16 low-traffic, 16 high-traffic trips	71.7 (low-traffic); 72.3 (high-traffic)	SD: 65.5 (low-traffic); 67.0 (high-traffic)

1. Method abbreviations: G=gravimetric, P=photometric
2. Due to the sometimes large local calibration factors applied to DustTrak readings, this attribute of the data processing was specifically sought in the papers and included in the table
3. Central values are arithmetic means unless otherwise noted (median, GM=geometric mean)
4. Variability is expressed as the range unless otherwise noted (SD=standard deviation, GSD=geometric standard deviation)
5. Additional data for Kingham et al. (2013) retrieved from Kingham et al. (2011)

Table S.6. On-road Measured Bicyclist Exposure Concentrations of PM_{10} (ug/m^3)

Reference	Location (setting)	Method[1]: Instruments	Modes (and/or bicycle routes) compared	Time frame	# on-bicycle samples	Central value [2]	Variability [3]
(Berghmans et al., 2009)	Mol, Belgium (small city)	P: Grimm 1.108	None	April 2007; weekdays (Mon-Thurs), 0600-1600	358 1-min samples	62.4	18.8-160
(Bernmark et al., 2006)	Stockholm, Sweden (urban)	P: MIE DataRam pDR-1000	None	Unspecified dates; weekdays; working hours	5 samples of 8-hr work-days	55	22-89
(Int Panis et al., 2010)	Brussels (B), Louvain-la-Neuve (L), and Mol (M), Belgium (small to large cities)	P: TSI DustTrak 8534	Auto (routes in 3 cities compared)	June-July 2009; weekdays; unspecified times	24 trips in B, 6 trips in L, 13 trips in M	63.4 (B); 50.0 (L); 72.7 (M)	SD: 17.0 (B); 7.6 (L); 13.4 (M)
(Jacobs et al., 2010)	Antwerp, Belgium (cycle track near a major roadway)	P: Grimm 1.108	None (clean room)	May 2009; unspecified days; 800-1700	38 trips	62.8	SD: 23.6
(Kingham et al., 2013) [4]	Christchurch, New Zealand (urban)	P: Grimm 1.101, 1.107, 1.108	Bus, auto (on-road and off-road routes)	February-March 2009; weekdays; 740-900 and 1645-1805	32 on-road, 46 off-road trips	Median: 32.0 (on-road); 35.3 (off-road)	12.9-61.7 (on-road); 8.2-91.4 (off-road)
(Nyhan et al., 2014)	Dublin, Ireland (urban)	P: Met One Aerocet 531	Walk, bus, train	Unspecified dates; weekdays; 800-1000	33 trips	55.2	SD: 30.1
(Rank et al., 2001) [5]	Copenhagen, Denmark (urban)	G: sample pump (1.9 L/min), Millipore filter (37 mm, 0.8 µm pore)	Auto	18 June and 3 August 1998; weekdays (Thurs and Mon); 740-940 and 1000-1200	4 samples of ~2 hr	44	21-68
(Strak et al., 2010)	Utrecht, Netherlands (urban)	G: Harvard impactor; sample pump (10 L/min), Pall PTFE filter (37 mm, 2-µm pore)	None (low-traffic and high-traffic routes)	April-May 2007; weekdays (Mon-Thurs); 800-930	14 low-traffic, 14 high-traffic trips	45.67 (low-traffic); 44.01 (high-traffic)	14.19-109.31 (low-traffic); 16.75-118.68 (high-traffic)
(Thai et al., 2008)	Vancouver, Canada (urban)	P: Grimm 1.108	None	August-October 2007; weekdays; 700-900	7 sample days	53.9	21.6-74.8
(Zuurbier et al., 2010)	Arnhem, Netherlands (urban)	G: Harvard impactor; sample pump (10 L/min), Pall PTFE filter (37 mm, 2-µm pore)	Bus, auto (low-traffic and high-traffic routes)	June 2007 – June 2008; weekdays (Tues and Thurs); 800-1000	15 low-traffic, 15 high-traffic trips	37.2 (low-traffic); 38.8 (high-traffic)	SD:11.6 (low-traffic); 14.4 (high-traffic)

1. Method abbreviations: G=gravimetric, P=photometric
2. Central values are arithmetic means unless otherwise noted (median)
3. Variability is expressed as the range unless otherwise noted (SD=standard deviation)
4. Additional data for Kingham et al. (2013) retrieved from Kingham et al. (2011)
5. Rank et al. (2001) report "total dust" without a PM size category; PM_{10} is assumed based on the instrumentation

Table S.7. On-road Measured Bicyclist Exposure Concentrations of EC, BC, and Soot (ug/m^3)

Reference	Location (setting)	Instruments	Modes (and/or bicycle routes) compared	Time frame	# on-bicycle samples	Central value [1,2]	Variability [2,3]
(Adams et al., 2002)	London, UK (urban)	HFPS – see Adams et al. (2001), EEL reflectometer with local calibration	Bus, rail, auto	July and August 1999, February 2000; weekdays; peak and off-peak, 4x/day	21 trips (July), 99 trips (August), 50 trips (February)	15.4 (July), 21.0 (August), 19.2 (February)	0.9-32.1(July), 1.6-48.4 (August), 4.8-62.8 (February)
(Dekoninck et al., 2013)	Ghent, Belgium (urban)	Magee portable aethelometer AE-51	None	Unspecified dates; weekdays (assumed); 730-930 and 1630-1830	209 trips	Median: 4 [5]	1-12
(Dons et al., 2012)	Flanders, Belgium (various environments)	Magee portable aethelometer AE-51	Walk, bus, light-rail, train, auto	Summer 2010 and Winter 2010-2011; all days; 24-hr sampling	1,167 5-min samples during bicycle travel	3.555	<1-10 [5]
(Hatzopoulou et al., 2013)	Montreal, Canada (urban)	Magee portable aethelometer AE-51	None	May-August 2011; weekdays (Mon-Thurs), AM: 800-1000 and PM: 1500-1700	57 trips (including AM and PM trips)	2.00 (AM); 1.05 (PM)	0.40-4.61 (AM); 0.20-2.51 (PM)
(Hong and Bae, 2012)	Seattle, USA (urban)	Magee portable aethelometer AE-51	None	June and September 2010; unspecified days; 730-900 and 1700-1830	Unspecified # of 1-min samples over 10 days	1.78	0.09-14.9
(Jarjour et al., 2013)	Berkeley, USA (urban, suburban)	Magee portable aethelometer AE-51	None (low-traffic and high-traffic routes)	April-June 2011; weekdays; 800-1000	9 low-traffic, 10 high-traffic trips	1.76 (low-traffic); 2.06 (high-traffic)	0.11-63.83 (low-traffic); 0.10-53.53 (high-traffic)
(Kingham et al., 1998) [4]	Huddersfield, UK (small city)	IOM inhalable dust sampler, smoke stain reflectometer; reported in units of Absorbance only	Bus, train, auto (on-road and cycle path routes)	September-October 1996; Mon-Sat; unspecified times	6 sample days	Absorbance: 2.7 (cycle path); 6.3 (on-road)	1.2-6.7 (cycle path); 2.9-15.1 (on-road)
(de Nazelle et al., 2012)	Barcelona, Spain (urban)	Magee portable aethelometer AE-51	Walk, bus, auto	May-June 2009; weekdays; 800-2000 (peak and off-peak)	34 trips	GM: 8.5	GSD: 1.7
(Nwokoro et al., 2012)	London, UK (urban)	Magee portable aethelometer AE-51	Non-bicyclists: walk and transit combined	November 2010-March 2011; weekdays; peak periods	14 sample days	11.681	SD: 1.375
(Strak et al., 2010)	Utrecht, Netherlands (urban)	Harvard impactors – see Table S.6, smoke stain reflectometer; reported in units of Absorbance	None (low-traffic and high-traffic routes)	April-May 2007; weekdays (Mon-Thurs); 800-930	16 low-traffic, 16 high-traffic trips	Absorbance: 4.35 (low-traffic); 6.03 (high-traffic)	1.07-13.96 (low-traffic); 2.31-16.03 (high-traffic)
(Weichenthal et al., 2011)	Ottawa, Canada (urban)	Magee portable aethelometer AE-51	None (low-traffic and high-traffic routes, indoors)	May-September 2010; weekdays; 1130-1230	40 low-traffic, 39 high-traffic trips	1.08 (low-traffic); 2.52 (high-traffic)	0.17-3.20 (low-traffic); 0.89-5.67 (high-traffic)
(Zuurbier et al., 2010)	Arnhem, Netherlands (urban)	Harvard impactors – see Table S.6, smoke stain reflectometer; reported in units of Absorbance	Bus, auto (low-traffic and high-traffic routes)	June 2007 – June 2008; weekdays (Tues and Thurs); 800-1000	16 low-traffic, 16 high-traffic trips	Absorbance: 5.3 (low-traffic); 6.6 (high-traffic)	SD: 2.8 (low-traffic); 3.2 (high-traffic)

1. Central values are arithmetic means unless otherwise noted (median, GM=geometric mean)
2. Three studies report BC in units of absorbance only (Kingham et al., 1998; Strak et al., 2010; Zuurbier et al., 2010)
3. Variability is expressed as the range unless otherwise noted (SD=standard deviation, GSD=geometric standard deviation)
4. Kingham et al. (1998) do not specify the type of PM; BC is assumed based on the instrumentation used (smoke stain reflectometer)
5. Values are extracted from a figure because values are unreported in text or tables

Table S.8. On-road Measured Bicyclist Exposure Concentrations of Other PM Sizes (ug/m^3)

Reference	Location (setting)	Method[1]: Instruments and PM size	Modes (and/or bicycle routes) compared	Time frame	# on-bicycle samples	Central value [2]	Variability [3]
(Berghmans et al., 2009)	Mol, Belgium (small city)	P: Grimm 1.108; PM_1	None	April 2007; weekdays (Mon-Thurs), 0600-1600	358 1-min samples	37.4	6.07-105
(Bevan et al., 1991)	Southampton, UK (urban, suburban)	G: sample pump (2 L/min), Millipore fluoropore filter; $PM_{3.5}$	None (urban and suburban routes)	Unspecified dates; weekdays (assumed); 800-900 and 1630-1730	16 trips	130	13-253
(Gee and Raper, 1999)	Manchester, UK (urban)	G: SKC sample pump (2.2 L/min) with cyclone head, Millipore fluoropore filter; PM_4	Bus	Unspecified dates; weekdays; 700-1000	8 samples of 3 hr	54	16.8-122
(Kingham et al., 2013) [4]	Christchurch, New Zealand (urban)	P: Grimm 1.101, 1.107, 1.108; PM_1	Bus, auto (on-road and off-road routes)	February-March 2009; weekdays; 740-900 and 1645-1805	32 on-road, 46 off-road trips	Median: 8.2 (on-road); 5.9 (off-road)	2.6-31.0 (on-road); 1.4-26.2 (off-road)
(Sitzmann et al., 1999)	London, UK (urban)	G: Casella sample pump (1.9 L/min) with cyclone head, Whatman glass fibre filter; PM_5	None	November 1995-February 1996; weekdays; peak periods	30 subjects, each with 5 samples of 1.5 hr; only 4 subjects reported	14.00, 16.28, 16.49, 88.54	SD: 2.34, 4.72, 4.07, 6.52
(Yu et al., 2012)	Shanghai, China (urban)	P: TSI DustTrak 8530 and 8533, local calibration by gravimetric analysis; PM_1	Walk, bus, subway, taxi (3 routes, peak and off-peak hours)	March 2011; weekdays; 730-1130	144 trips	140	SD: 86

1. Method abbreviations: G=gravimetric, P=photometric
2. Central values are arithmetic means unless otherwise noted (median)
3. Variability is expressed as the range unless otherwise noted (SD=standard deviation)
4. Additional data for Kingham et al. (2013) retrieved from Kingham et al. (2011)

4 BICYCLIST RESPIRATION MEASUREMENTS

Table 2 in the paper summarizes published traveling bicyclists' respiration parameters. Some of the studies in Table 2 measured respiration on-road, while others used bicycle ergometer laboratory testing; see Weisman (2003) for a discussion of physiology and exercise testing. Without measuring on-road workloads it is difficult to compare the conditions of the laboratory tests with on-road bicycling. McNabola et al. (2007) found a linear relationship between speed and V_E based on ergometer testing, while Adams (1993) found nonlinearly increasing ventilation with bicycling speed. The difference in results could be explained by the laboratory setting neglecting the strong effects of aerodynamic drag on increasing workload with bicycling speed (Faria et al., 2005).

A third study methodology in Table 2 combines laboratory bicycle ergometer tests with on-road heart rate (HR) monitoring to estimate on-road respiration. This method relies on the strong intra-subject relationship between HR and (log-transformed) V_E for bicycling (Samet et al., 1993; Zuurbier et al., 2009) and is appealing because HR is easier to measure *in situ* than V_E. Consistent with the ranges in Table 2, Mermier et al. (1993) observed average V_E of around 15 to 60 L/min for laboratory bicycling exercise tests with HR from 80 to 140 beats per minute (bpm). Comparing the slope estimates for $\ln(V_E)$ as a function of HR (in L/min and bpm) while bicycling, the results in Zuurbier et al. (2009) and Mermier et al. (1993) agree well, with group means in the range of 0.019 to 0.023 for healthy subjects (men, women, boys, and girls). Bernmark et al. (2006) do not report their estimated slopes, but an example figure shows a slope of 0.018.

Energy expenditure is a key factor for respiration and thus air pollution intake (Nadeau et al., 2006). Creating external work requires delivery of oxygen to body tissues, which in turn requires inhalation of oxygen. The volume rate of oxygen inhalation (VO_2), which is closely related to V_E, increases "nearly linearly" with external workload or power (Weisman, 2003). For this reason, Vinzents et al. (2005) use a slightly different approach from the "estimated" method in Table 2 to model pollution intake, establishing individual HR-workload relationships using a bicycle ergometer and monitoring on-road HR to estimate workload during travel, which they assume is linearly proportional to V_E.

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