The Impact of Route Choice on Active Commuters' Exposure to Air Pollution: A Systematic Review

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As air pollution varies significantly in both space and time, commuter exposure may differ considerably depending on the route taken between home to work. This is especially the case for active mode commuters who often have a wider range of route choices available to them compared with those traveling by private motor vehicle or by public transport. The aim of this study was to investigate the effect of route choice on air pollution exposure among active commuters, and to estimate, based on modeling, the health benefits able to be achieved from air pollution exposure reductions, modeled across a population, through route optimization. We searched for studies that used portable personal air pollution monitoring equipment during active mode commuting, and reported measurements of air quality on at least two routes, either as a journey to work or to school. The World Health Organization (WHO) model AirQ+ was then exploited to estimate the premature deaths attributable to air pollution according to route choice. Ten publications were identified that met the inclusion criteria. Ultrafine particle counts (UFP), black carbon (BC), and carbon monoxide (CO) were the most commonly measured pollutants in the studies identified. The exposures associated with "high exposure" and "low exposure" routes (categorized based on differences in traffic counts on the roads along the commute route or walking on opposite sides of the road with different levels of traffic traveling in each direction) were found to vary on average by $30 \pm 8\%$, $42 \pm 35\%$, and $55 \pm 17\%$ for BC, CO and UFP, respectively. On the basis of modeling, and on the estimated exposures to BC, up to 36 out of 10,000 deaths could be prevented by choosing a low exposure route compared with a high exposure route during active commuting. The results of this study may be useful for both individuals in their commute planning, and also for urban transport planners as impetus for investing in infrastructure to support healthy active mode commuting.

Keywords: air pollution, route choice, health outcome, active commuters, systematic review

INTRODUCTION

Over the past half a century, urbanization and the reliance on passive transport (especially private cars) have led to low levels of physical activity across the world (World Health Organization, 2020c), which is a major risk factor for mortality (World Health Organization, 2010). More than 1.4 billion people of the world's population are deemed to not be sufficiently active to be healthy (Thornton, 2018), with insufficient physical activity contributing to a higher risk of cardiovascular disease, Type 2 diabetes, dementia, and some cancers (Guthold et al., 2018).

The promotion of active transportation (AT) is one of the strategies that has been used by governments to help address the issue of inactivity (Goodin, 2015). While AT has traditionally been considered to be walking, running and cycling, this category includes any type of transport powered by human energy (Litman, 2018). AT can, to a large extent, provide sufficient physical activity to remain healthy (Shoham et al., 2015) and, if rolled out at scale, worldwide, it could potentially prevent 3.2 million deaths due to inactivity (Guthold et al., 2018). AT is also associated with environmental improvements, and has been shown to also have social and cultural benefits (Hong, 2018). For instance, parents who choose active transportation influence their children to also be active, and employees using active transport have been shown to help in encouraging co-workers to change their choice of mode of transport to ones that are more active (Hong, 2018).

Active commuting is also a more environmentally sustainable mode of transport than the use of private vehicles as it requires little in the way of fossil fuels or cost (Banister, 2009; Page and Nilsson, 2017; Hong, 2018). Private cars also contribute more to soil and water pollution through the production of vehicle spray, and the dry deposition of particles from vehicles and roads dispersed into roads' verges and surface water (Schipper et al., 2007; Maibach et al., 2008; Hong, 2018).

Despite the important advantages of AT, there are some associated risks, with the fear of traffic injury being a common reason given by many for their preference for motorized transport over walking or cycling (Sonkin et al., 2006). While this perception is not unfounded [the risk of hospital admission due to traffic related crashes is higher for walkers and cyclists than for drivers (Mindell et al., 2012)], the risk is none-the-less low in comparison with many activities such as do-it-yourself repairs (DIY) at home, as well as common recreational activities (Chieng et al., 2017).

The risk that is often not taken into account in active transportation is the health impact caused by exposure to air pollution (Bigazzi and Figliozzi, 2014; Dirks et al., 2018). Active travelers tend to spend more time than passive commuters traveling the same distance, and experience higher minute ventilation, potentially increasing the net dose of air pollution received by the body and consequently, for the same origin/destination pair and the same route, active commuters may experience a higher dose of pollution compared with passive commuters (Bigazzi and Figliozzi, 2014; Cepeda et al., 2017; de Nazelle et al., 2017; Targino et al., 2018).

One approach that has been used to reduce active commuters' exposure and promote active commuting is through changes in the urban design of cities, such as increasing the separation between commuters and motorized road traffic. For example, in one study, it was found that a separation of 100 meters decreased ultrafine particle counts (or UFPs which are particles smaller than 100 nanometers in diameter), black carbon (BC) and carbon monoxide (CO) concentrations relative to the roadside by 60 to 80 percent (Zhu et al., 2002). Providing such a separated path has also been found to result in the public perception of active transport as safe from the point of view of road traffic injury (Hull and O'Holleran, 2014). Thus, creating the appropriate transport infrastructure for active commuters can encourage people to shift from private cars to active commuting (Caulfield et al., 2012). Another strategy can be to limit road traffic, particularly HCVs (heavy commercial vehicles including buses and trucks) in specific areas, and to redesignate these as active commuter areas, as shown by Mueller et al. (2020).

In the absence of any information regarding air pollution exposure and route choice, active commuters tend to choose routes that involve the least number of directional changes or the shortest distance (Shatu et al., 2019). Given that the shortest route is not always the one associated with the lowest air pollution dose (Armeni and Chorianopoulos, 2013), the provision of information regarding air pollution and traffic could help commuters shift toward lower exposure routes (Bunds et al., 2019). The information could be provided using either traditional environmental monitoring equipment or low-cost sensors with community involvement provided at a grass-roots level (Mahajan et al., 2020). As such, governments may exploit citizen science (a process in which people with potentially minimal relevant experience become involved in data collection) to provide fine-grained information using mobile apps and low-cost sensors (Mahajan et al., 2020). In such a case, academic scientists are often responsible for implementing and interpreting the results (Eitzel et al., 2017). In this case, this would be in relation to identifying the least polluted routes, and estimates of health gains, for example, that could be achieved by using such routes.

Exposure assessment at the individual level is a key tool for assessing health impacts and developing suitable exposure-reduction policies. Traditionally, estimates have been made based on fixed-site monitoring, supplemented by exposure modeling, albeit with a high degree of uncertainty due to the high spatial and temporal variability of air pollution levels (Dias and Tchepel, 2018). A more reliable and accurate way of estimating personal exposure is through the use of personal portable air pollution monitoring devices in combination with a Global Position System (GPS) to track location (Dias and Tchepel, 2018). Consequently, a large and growing body of literature has explored individual's exposure using such devices (Strak et al., 2010; Jarjour et al., 2013; Dirks et al., 2016, 2018; Good et al., 2016; Pattinson et al., 2017; Hofman et al., 2018; Jereb et al., 2018; Brand et al., 2019; Luengo-Oroz and Reis, 2019). While such studies provide, at the individual level, reliable exposure information, they do not generally link directly to health outcomes. In particular, there is a lack of research into the health benefits that could be achieved though route optimization.

In recent years, there has been a growing interest in the use of health impact assessments (HIA) to estimate the potential benefits to health associated with various interventions (Mueller et al., 2015). Many models have been developed for such a purpose, including those published by the European Study of Cohorts for Air Pollution Effects (ESCAPE) project and the American Cancer Society (ACS) (Malmqvist et al., 2018). The World Health Organization (WHO) model AirQ+ (World Health Organization, 2020b) is designed to allow estimation of both the long- and short-term health impacts of exposure to different pollutants based on a concentration-response function extracted from epidemiological studies. In light of the many

health effects attributed to combustion-related BC, including all-cause and cardiopulmonary mortality (Janssen et al., 2012; Kirrane et al., 2019), this study links evidence from the literature about the reduction in exposure to BC able to be achieved through route optimization, and uses an HIA approach to estimate the associated predicted human health benefits.

This paper begins with a systematic review of measurement studies that have used portable air quality monitoring equipment to investigate differences in air pollution exposure experienced by active mode commuters' exposure, taking different routes for the same origin-destination pair. The extent of health benefit that may be able to be achieved through route optimization is then estimated using AirQ+ software.

MATERIALS AND METHODS

Systematic Review

The systematic review was performed based on the PRISMA guidelines for the reporting of systematic reviews (Moher et al., 2009). The search was conducted in October 2019 using the following databases: MEDLINE, Web of Science, and Scopus. The following keywords were used in the search:

("route choice" or (route AND optimi*) OR "short* path") AND ("air pollut*" OR "air quality" OR emission* OR "traffic pollut*" OR (pollut* AND exposure)) AND (walk* OR bicycl* OR cyclist OR cycling OR bik* OR pedestrian OR "active commut*" OR "active transport*")

The search was restricted to publications written in the English language and that included an abstract. The eligible papers also had to meet all of the following criteria:

1. Air pollution data were measured directly (not modeled) using portable air pollution monitors.
2. Air pollution exposure was measured for at least two routes for the same origin and destination pair.
3. Quantitative results of dose/exposure were reported for the time when the routes were traversed.
4. The percentage reduction achieved by taking the lowest exposure route compared to the highest exposure route was reported or able to be calculated based on the figures provided.

The study data collected included air pollution exposure, sample size, location, time and date of sampling, and target group (children or adults). The literature search, study selection, data extraction, and synthesis were performed between February and March 2020. The primary outcome measure was the percentage difference in exposure between the highest and lowest exposure route. The mean exposures for the high and low exposure routes were extracted. If the study involved more than two routes, the route with the highest exposure was selected and compared with the route with the lowest exposure.

Estimate of Health Effects

Based on the results of the systematic review, the long-term health benefits, able to be achieved by taking the low exposure route rather than the high exposure route, were estimated. This was based on the predicted health effects of exposure to BC for a commuter who travels

the same route every working day for a year. All-cause mortality was chosen as it is an important public health benchmark, providing more robust results than cause-specific mortality and morbidity (Rojas-Rueda et al., 2016). The proportion of deaths attributable to BC was also able to be provided by AirQ+ on the basis of limited data input requirements.

To estimate the annual mean exposure to air pollution, a series of assumptions were made. Firstly, it was assumed that a person spends 1 h per day [which is the average time of daily travel to work for many countries (Silveira Neto et al., 2014; Perry, 2016)] over 236 days of a year in the transport microenvironment (assuming 25 days of holidays and 104 weekend days), and that a person is exposed to background levels of BC at other times. The mean exposures for the person commuting the high exposure route and the low exposure route were calculated for each of the reviewed studies. By dividing the difference by 24, the daily difference (for a weekday) of exposure was calculated. This number was then multiplied by the fraction of working days over a year to get the yearly reduction in the long-term exposure to BC for a commuter who consistently follows the low-exposure route compared with the high exposure route. The background concentration was calculate by assuming (arbitrarily) the ratio of the background to the average concentration along the low exposure route to be 1, 0.5, 0.33, 0.25, and 0.20. On this basis, the annual exposures for both the high and low exposure routes were calculated for each of the studies identified in the systematic review. The final output is the annual mean predicted BC exposure for a given route.

To estimate the long-term health effects of exposure to BC, AirQ+ was used (World Health Organization, 2020b). This model employs concentration-response functions derived from a systematic review of epidemiological studies until 2013 (World Health Organization, 2020b). In contrast with other models, such as ACS and ESCAPE, each of which was developed for a specific region and for a limited number of pollutants, AirQ+ includes BC (which is not included in the other models) and is considered to be applicable across a range of countries, including those in Western Europe and in North America (World Health Organization, 2020b).

To quantify the attributable proportion of all-cause mortality using AirQ+, the model requires a cut-off value of concentration (we considered this to be zero) and air quality data, including long-term exposure to BC (above the cut-off). The attributable proportion is given by Ansari and Ehrampoush (2019) as

$$AP = [(RR(c) - 1)^*p(c)]/[RR(c)^*p(c)]$$

where RR(c) is the relative risk for the health endpoint in a specific category of exposure (c), and p(c) is the proportion of the population in the category of exposure (c). For calculating the RR, the default values provided by AirQ+ were used, and a linear-log method was exploited to estimate the relative risks associated with different exposures. In this study, the differences in the estimated attributable proportion of deaths for high and low exposure routes have been considered to be the contribution

to mortality reduction of the least polluted commuting route compared with the most polluted route.

RESULTS

This section is divided in three subsections. The first part explains the results of the search. Then, a description of the extent of reduction in exposure to pollutants found in the searched studies is provided. Finally, a health impact assessment approach, used to estimate the health benefit able to be achieved through route optimization for air pollution exposure, is presented.

Search Result

In total, 333 studies were identified with 25, 72, and 233 from Medline, Scopus, and the Web of Science, respectively, and another three studies identified from other sources. Of the 333 studies, 24 were duplicates and were thus removed. Additionally, MR screened the titles manually and deleted 31 more studies because of duplication. After screening the title and abstract, 18 papers were considered to meet our inclusion criteria. The full
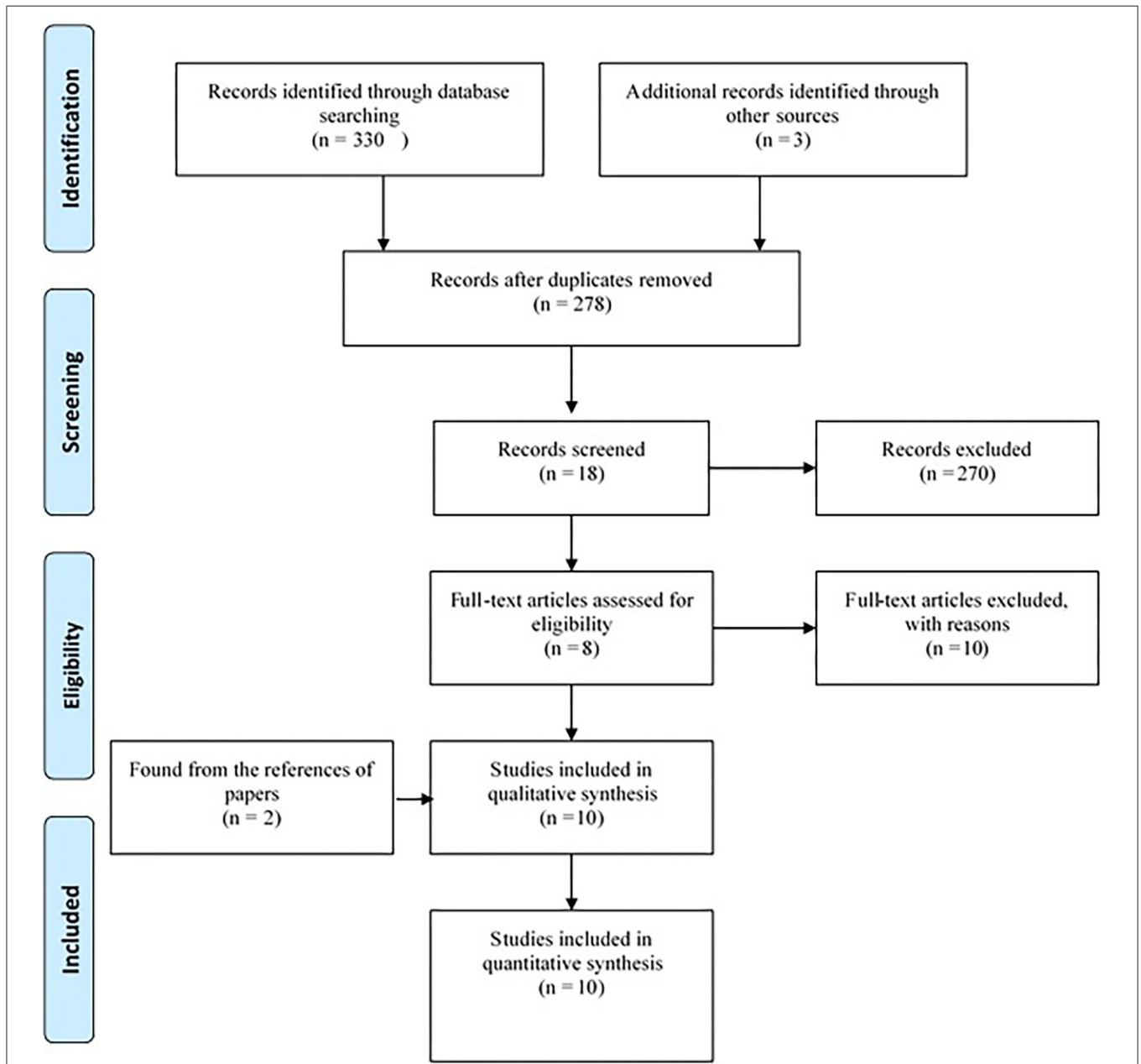

FIGURE 1 | Flow chart of information through the different phases of a systematic review.

texts were then analyzed to assess for eligibility. Eight papers were considered to be eligible according to our criteria. Two more studies were included through cross-references, leaving a total of 10 studies for the systematic review (**Figure 1**).

Table 1 shows the main characteristics of the included studies, such as the pollutant of interest, the mode of travel, the country and the city in which data collection took place, the population density of the city, the study group, the mean and standard deviation of the exposure for each route, the number and time of day of the trips, and the mean percentage difference associated with the low exposure route compared with the high exposure route.

Exposure Reduction

The most commonly measured pollutants were UFP (seven studies), BC (five studies), and CO (four studies). However, other pollutants, including PM2.5 and PM1, (particulate matter of <2.5 and 1 micrometers in diameter) were also measured. The mean ratios of high-to-low exposure were found to be 1.30 ± 0.08, 1.55 ± 0.16 and 1.42 ± 0.035 for BC, UFP and CO, respectively. In each of the studies, two routes were compared with respect to the active commuters' exposure to air pollution. One of the routes was through high traffic roads (or on the high traffic side of the road), and the other route was either away from traffic or on the low traffic side of the road. The percentage reduction in exposure between routes varied depending on the type of pollutant, the time of day, data collection location and the device used.

Figure 2 shows concentration differences in UFP ranging from 7,990 to 44,090 pt/cc, depending on the device used and the study setting. By reviewing the dates of the data collection for the different studies measuring UFP exposure, it was found that older studies tended to result in higher mean exposures compared with the more recent studies. For example, the highest mean UFP exposure was observed in the study by Strak et al. (2010) in which data collection took place in April 2007, while the second highest exposure to UFP was measured in Christchurch by Pattinson et al. (2017) based on data collected in 2009. However, in the other studies, the UFP concentration in the high exposure route was found to be close to 20,000 pt/cc or less.

BC was measured in four of the studies. The study by Brand et al. (2019) measured BC at three different cities (**Figure 3**). In this study, the mean reduction in exposure between routes was found to be 1.21 (SD = 0.97) $\mu g/m^3$. The highest values of BC were observed by Brand et al. in Sao Paulo and the lowest in Rotterdam (Brand et al., 2019). Furthermore, the highest reduction in exposure to BC achieved by choosing the low exposure route compared to the highest route was found in the study conducted in Sao Paulo (Brand et al., 2019).

Four studies measured CO. One of these reported only the percentage difference in the mean exposure (−10%, CI=-20 ± 1), which was found to be not significant. The other three studies provided the numerical means for each route (**Figure 4**). The mean reduction was found to be 0.23 (SD = 0.31) ppm. The highest ratio of reduction was found in a study by Pattinson et al. (2017) based on measurements made in 2009. This suggests that higher levels of concentration tend to be recorded in older

studies carried at a time when the vehicle fleet technology was less advanced.

In some studies, other pollutants were measured, including soot, PM2.5, PM10, and PM1. In two studies, PM2.5 was considered (Jarjour et al., 2013; Good et al., 2016). In both studies, the reduction in mean exposure between routes was found to be not significant. Strak et al. reported 8.10 $\mu g/m^3$ and 1.68 $\times 10^{-5}$/m reductions in cyclist's exposure to PM10 and soot, respectively. One study considered PM1 (Pattinson et al., 2017) and observed a decrease of 1 $\mu g/m^3$.

In these studies, various instruments (with different settings) were used for the measurement of pollutants. For example, three different devices [the CPC TSI 3007 (Strak et al., 2010; Jarjour et al., 2013; Pattinson et al., 2017), P-Trak (Dirks et al., 2016, 2018; Hofman et al., 2018), DiscMini (Luengo-Oroz and Reis, 2019), and MiniDiSC (Good et al., 2016) were used to measure UFP numbers for particle sizes in the range of 0.01 to 1 μm, 0.02 to 1 μm (Matson et al., 2004), 0.01 to 0.3 μm (Testo SE and Co, 2019), 0.01 to 0.3 μm (NANEOS, 2020)]. All of the studies presented in this review used an AE51 for measuring black carbon (Jarjour et al., 2013; Hofman et al., 2018; Brand et al., 2019), except for one (Jereb et al., 2018) which used an AE33. The aethalometer AE51 measures only BC at a wavelength of 880 nm, and is used to detect black carbon. An AE33 can be operated using various different wavelengths. However, in the study by Jereb et al. (2018), a pair of wavelengths, 470 and 950 nm, was used to measure traffic-related BC. In some studies, soot and BC were considered to be the same, although there are fundamental differences between them (Long et al., 2013). Optical methods were used to measure BC but for the quantification of EC, Strak et al. (2010) post-processed PM10 filters using a Smoke Stain Reflectometer (model M43D; Diffusion Systems, London, UK) and transformed the results into absorbance. For evaluating CO, two instruments were used: a Langan (Good et al., 2016; Dirks et al., 2018) and a Qtrak (Jarjour et al., 2013).

Of the ten studies included in the review, five compared either morning and afternoon or evening exposures (Dirks et al., 2016, 2018; Hofman et al., 2018; Jereb et al., 2018; Brand et al., 2019). Of the five studies, three considered UFP, one considered CO and three considered BC. For the studies that considered UFP, the ratios of the morning-to-afternoon concentrations were found to be 2.80, 1.30, and 1.65 for Auckland, New Zealand (Dirks et al., 2018), Bradford, UK (Dirks et al., 2016) and Antwerp, Belgium (Hofman et al., 2018), respectively. The one study (Auckland) that measured CO (as well as UFP), reported a ratio of morning to afternoon mean concentration of 1.04 (Dirks et al., 2018). Mixed results were found for BC. It was shown that in Sao Paulo, Brazil, Rotterdam, The Netherlands, and Antwerp, Belgium, the mean morning exposures to BC were 49, 13, and 33% higher than in the evening, respectively. The morning-to-afternoon black carbon levels measured in Celje, Slovenia were found to be 0.94 and 1.12 on the main and alternative routes, respectively. Brand et al. (2019) also observed lower concentrations in the morning relative to the evening, with a ratio of 0.93. Therefore, although the differences in exposure between morning and afternoon or evening varied, most studies found average exposures to be
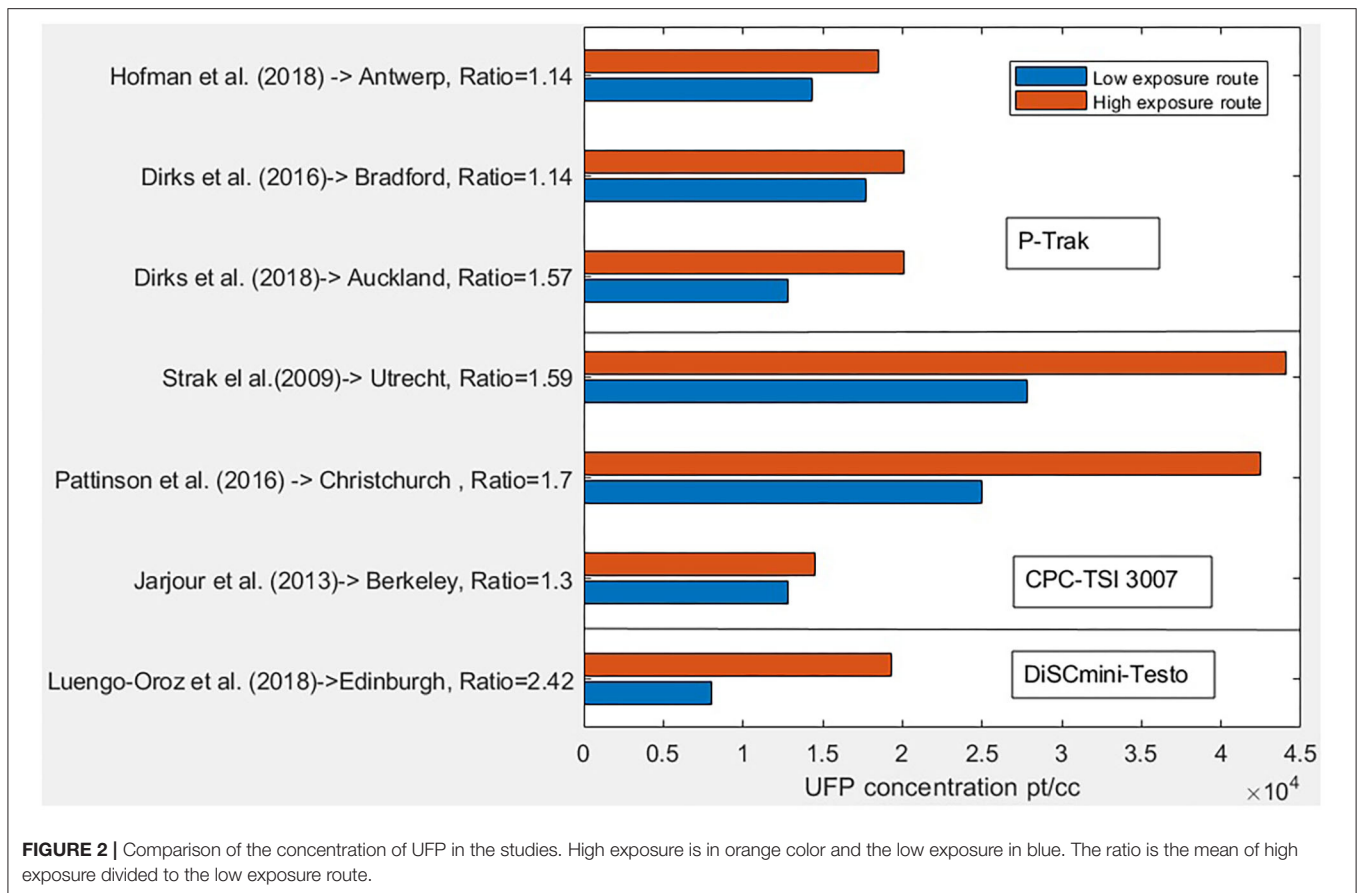
TABLE 1 | Main characteristics of the studies included in the systematic review (Int = interval, Pop den = Population density, cutoff: the threshold for the size of particles).

References	Modes	Pollutant	Location	Equipment	Group	Difference in exposure (%)	Mean (SD)	Trips
Dirks et al. (2018)	Walking	CO UFP	Auckland, NZ	UFP: P-Trak (8525 TSI incorporation, US, cutoff: 20 nm) CO: Langan (Langans Products, Inc,UK)	Children	a.m.: UFP: 57 CO: 0 p.m.: UFP: −0.02 CO:0.08	a.m.: UFP (pt/cc) R1: 20,100 (1,300) R2: 12,800 (900) CO (ppm) R1=R2: 0.73 (0.07) p.m.: UFP (pt/cc) R1: 5,800 (1,000) R2: 5,900 (900) CO (ppm) R1: 0.42 (0.11) R2: 0.39(0.32)	18 (9 north + 9 south) a.m. 6 (3 north + 3 south) p.m. 10/2015(Spring)
Dirks et al. (2016)	Walking	UFP	Bradford, UK	UFP: P-Trak (8,525 TSI incorporation, US, cutoff:20 nm)	Children	a.m. UFP: 14 p.m. UFP:2	a.m.: R1: 17,680 (21,060) (pt/cc) R2: 20,100 (22,000) (pt/cc) p.m.: R1: 14,716 (19,683) (pt/cc) R2: 14,383 (17,016)(pt/cc)	5 trips a.m. 5 trips p.m. 11/2015(Winter)
Luengo-Oroz and Reis (2019)	Cycling	UFP	Edinburgh, Scotland	UFP: DiSCmini (Testo SE & Co. KGaA, Germany, cutoff: 10 nm)	Adults	UFP: 97	R1: 9,824 pt/cc R2: 7,990 pt/cc R3: 19,310pt/cc	3 routes * 9 days * 2 (onward and return) a.m. 06/2018 to 07/2018(Spring)
Jarjour et al. (2013)	Cyclist	UFP CO PM$_{2.5}$ BC	Berkeley California, US	UFP: CPC (model 3,007; TSI Inc., US, cutoff: 10 nm) CO: Q-Trak (7,565, TSI Inc., US) PM$_{2.5}$: DustTrak (8,520, TSI Inc., US) BC: microAeth® (AE51, TSI Inc., US, 880 nmwavelength)	Adults	UFP: 30 CO: 14 PM$_{2.5}$: 0.05 BC:0.15	UFP (pt/cc) R1: 14,311 ± 15,381 R2: 18,545 ± 42,482 CO (ppm) R1: 0.79 ± 0.39 R2: 0.90 ± 0.64 PM$_{2.5}$ ($\mu g/m^3$): R1: 4.88 ± 1.41 R2: 5.12 ± 1.86 BC R1: 1.76 ± 2.58 R1: 2.06 ±3.23	9 (sampling days) *2 (routes)* 15 (people) a.m. 04/ 2011 to 06 / 2011 (Spring: dry season inBerkeley)
Strak et al. (2010)	Cycling	PM$_{10}$ soot UFP	Utrecht Netherlands	PM$_{10}$ Indoor pumps (model SP-280E; Air Diagnostics and Engineering, Harrison, ME, US) Soot Smoke Stain Reflectometer (model M43D; Diffusion Systems, London, UK) UFP: CPC (model 3007; TSI Inc., US, cutoff:10 nm)	Adults	PM$_{10}$:0.04 UFP: 59 Soot:0.39	PM$_{10}$ ($\mu g/m^3$): R1: 45.67 (28.35) R2: 44.01 (29.39) UFP (pt/cc) R1: 27,813 (5,919) R2: 44,090 (10,036) Soot ($10^{-5}/m$): R1: 4.35 (3.19) R2: 6.03(3.53)	12 days for UFP a.m. 04/2007 to 05/2007(Spring)

(Continued)

TABLE 1 | Continued

References	Modes	Pollutant	Location	Equipment	Group	Difference in exposure (%)	Mean (SD)	Trips
Jereb et al. (2018)	Cycling	BC	Celje/Slovenia	Magee Scientific Aethalometer (AE33, Aerosol d.o.o., Ljubljana, Slovenia, a pair of 470 and 950 nm wavelengths were used)	Adults	a.m. BC: 13 p.m. BC:42	Morning R1: 6.0 ± 5.1 $\mu g/m^3$ R2: 5.3 ± 4.8 $\mu g/m^3$ Afternoon R1: 6.4 ± 4.9 $\mu g/m^3$ R2: $4.5 \pm 6.0 \mu g/m^3$	One day: Twice in the morning Twice in afternoon 05/2017(Spring)
Hofman et al. (2018)	Cycling	BC UFP	Antwerp/Belgium	BC: microAeth® (AE51, TSI Inc., US, 880 nm wavelength) UFP: P-Trak (8,525 TSI incorporation, US, cutoff:20 nm)	Adults	BC:156 UFP:14	BC ($\mu g/m^3$) R1: 3.38 R2: 1.32 UFP (pt/cc) R1: 14553 R2:12818	10 days twice morning and twice afternoon then all measured data were averaged along each route. 01/2017 to 03/2017(Winter)
Brand et al. (2019)	Cycling	BC	London/UK Rotterdam/Netherland SaoPaulo/Brazil	AE51 microAeth® (AE51, TSI Inc., US, 880 nm wavelength)	Adults	London: 48 Rotterdam: 6 Sao Paulo:52	London R1: 5.53 $\mu g/m^3$ R2: 3.73 $\mu g/m^3$ Rotterdam: R1: 1.63 $\mu g/m^3$ R2: 1.73 $\mu g/m^3$ Sao Paulo: R1: 7.23 $\mu g/m^3$ R2: 4.76$\mu g/m^3$	Morning Afternoon Evening 8-12 trips Rotterdam (10/2017) Autumn London (12/2017) Winter Sao Paulo (02/2018)Summer
Pattinson et al. (2017)	Cycling	UFP CO PM1	Christchurch/NZ	UFP: CPC (model 3007; TSI Inc., US, cutoff: 10 nm) CO: Langan (Langans Products, Inc, UK) PM$_1$: Dust monitor (Model 1.107, GRIMM Aerosol Technik,Germany)	Adults	R3 to R1 UFP: 70 CO: 111 PM1:24	UFP (pt/cc) R1: $42,480 \pm 27,176$ R2: $28,765 \pm 18,638$ R3: $24,977 \pm 19,650$ CO (ppm) R1: 1.12 ± 0.59 R2: 0.89 ± 069 R3: 0.53 ± 0.39 PM$_1$ ($\mu g/m^3$) R1: 5.1 ± 2.9 R2: 5 ± 2.6 R3: 4.1 ± 2.1	5 days sampling afternoon various distances to road (03/2009 to 04/2009)Autumn
Good et al. (2016)	Cycling	BC CO PM$_{2.5}$ UFP	Collins, US	PM$_{2.5}$:PEM, SKC BC: microAeth® (AE51, TSI Inc., US, 880 nm wavelength) UFP: DiSC Mini (Matter Aerosol AG, Testo Inc, Switzerland, Cutoff: 1,000 nm) CO: Langan (Langans Products, Inc,UK)	Adults	BC (mean) -23 $(-32, -13)$ BC (cumulative) $-35 (-51, -12)$ CO (mean) -10 $(-20, +1)$ CO (cumulative) $-30 (-46, -9)$ (mean) $-7 (-20, +9)$ PM$_{2.5}$ (cumulative) $+3 (-18, +28)$ UFP (mean) $+1$ $(-35, +59)$ UFP (cumulative) $+17 (-29,+93)$	–	381 days of data were collected, including 678 valid commutes (350 morning and 328 evening), 8 commute days for each participant 11/ 2012 to 02/ 2014 Winter

FIGURE 2 | Comparison of the concentration of UFP in the studies. High exposure is in orange color and the low exposure in blue. The ratio is the mean of high exposure divided to the low exposure route.

higher during the morning rush hour than at other times of the day.
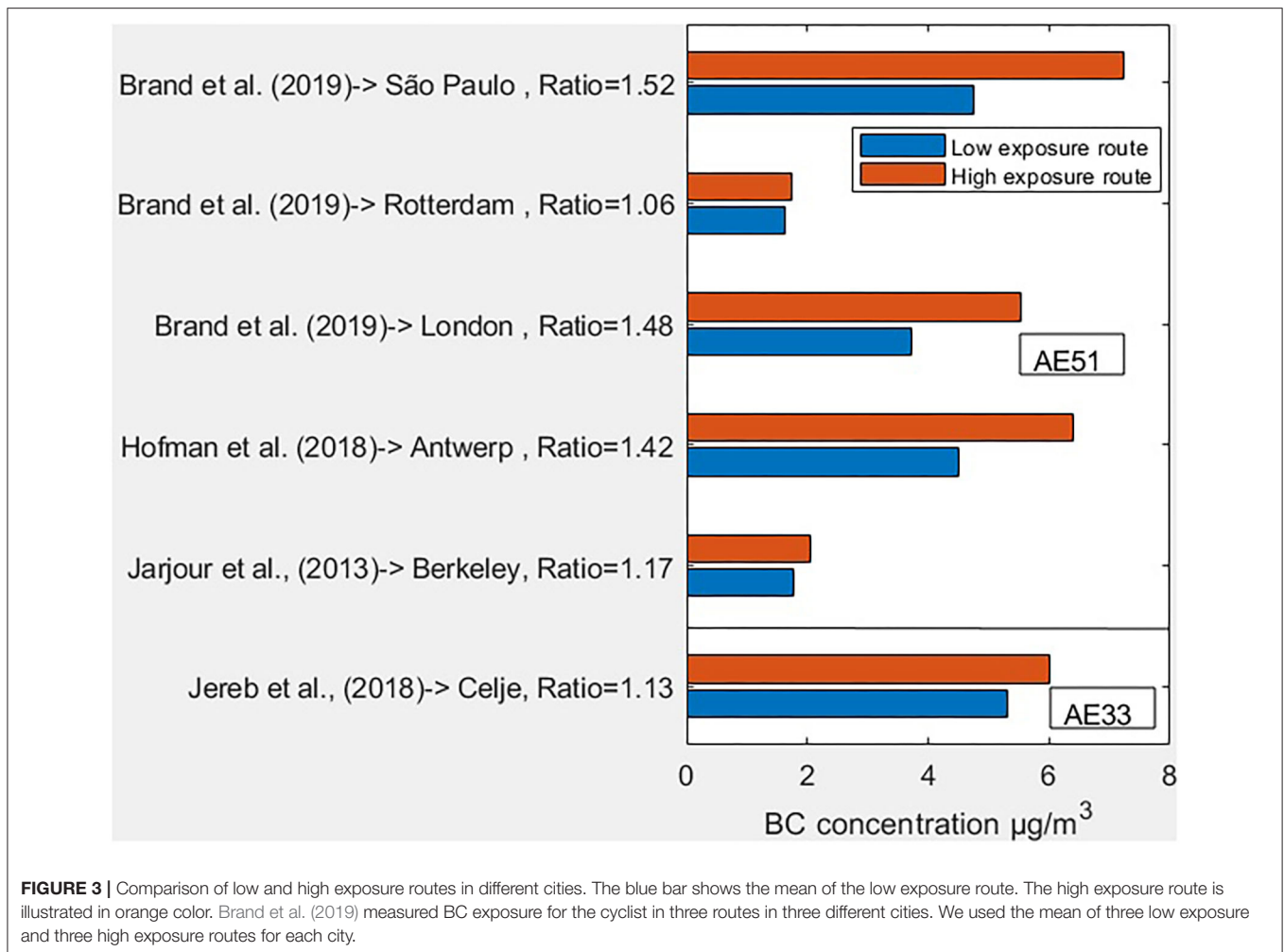
Long-Term Health Outcomes

Using AirQ+, the Estimated Attributable Proportion of the all-cause mortality– fraction of mortality attributed to exposure to black carbon for both the high and low exposure routes were calculated based on the average commuting exposures, along with the differences between the two routes. Four studies provided cyclists' exposure to BC in six different cities. Based on the data presented in the studies reviewed, the mean exposures and background concentrations were calculated based on the measurements made for the different cities, and for when the ratios of the background concentration to the low exposure route are 1, 2, 3, 4, and 5, as presented in **Table 2**. Given that the background concentrations are not the same around the world [some cities (including Auckland) tend to have cleaner air, while many cities have much worse conditions], the five different background concentrations for each city were calculated by dividing the low exposure route to 1, 2, 3, 4, and 5 to account for the variability in background concentration. The annual exposure was computed by choosing low and high exposure routes and considering various background exposures (**Table 3**). The results of the model output from AirQ+ are presented in **Table 4**. Note that the differences in the attributable proportion

of mortality is affected significantly by route choice, though the levels of background concentration also have an influence, with the relative difference being smaller in conditions of high levels of background. On average across cities, the difference in attributed proportions was found to be 0.17%, indicating that, on average, 17 of 10,000 deaths (of the people biking 1 h per day to and from their work) could be prevented by taking a lower exposure commuting route. In Sao Paulo, where the highest difference in exposure between routes was recorded, on the basis of this methodology, up to 36 in 10,000 deaths may be able to be prevented (Brand et al., 2019).

DISCUSSION

This systematic review compiled papers investigating the extent to which reductions in air pollution exposure could be achieved by the selection of lower exposure alternative routes by active mode commuters. The review showed that, generally, using alternative routes significantly decreases active commuters' exposure to air pollution, with average reductions of 55, 42, and 30 percent for UFP, BC, and CO, respectively. The extent of the reductions varied depending on the nature of the alternative routes considered, with more significant reductions observed for those commuting through green spaces, for example, where the route involved significant separation from the road.

FIGURE 3 | Comparison of low and high exposure routes in different cities. The blue bar shows the mean of the low exposure route. The high exposure route is illustrated in orange color. Brand et al. (2019) measured BC exposure for the cyclist in three routes in three different cities. We used the mean of three low exposure and three high exposure routes for each city.

The measured exposure depends on many factors which can vary from one pollutant to another. These include the instruments used for capturing pollutant concentrations, the time of day, the season and the year of data collection, the fleet composition, and the population density. Furthermore, factors such as passing construction sites, intersections and heavy duty vehicles were shown to lead to spikes in exposure to UFP whilst active traveling (Dirks et al., 2018; Luengo-Oroz and Reis, 2019). Significant reductions in exposure can thus be achieved by avoiding these sources by, perhaps temporary, changes in routes.

While CO is mostly associated with emission exhaust from petrol cars (Kingham et al., 2013), diesel vehicles such as trucks and buses are responsible for high rates of emissions of UFP (Luengo-Oroz and Reis, 2019) and BC (Longley et al., 2019); Heavy-duty vehicles, generally powered using diesel fuel, emit higher amounts of BC (Kumar et al., 2018) and UFP (Knibbs et al., 2011) per vehicle than petrol cars. Note that the ratio of the emission factors between heavy-duty vehicles and light-duty vehicles has been estimated to be about four for BC (de Miranda et al., 2019) and up to 40 for UFP (Xiang et al., 2019). High traffic roads tend to have larger numbers of heavy-duty vehicles than

roads with relatively light traffic (i.e., low numbers of vehicles of all kinds) (Jarjour et al., 2013).

The estimation of the long-term effect of exposure to black carbon in relation to route choice showed that, based on the average exposure of all studies, 17 of 10,000 deaths would be able to be prevented if active mode commuters took the associated low exposure route rather than the high exposure route for their journey to work. Based on the highest difference between the low and high exposure routes, which was recorded in Sao Paulo, it is estimated that up to 36 in 10,000 deaths are able to be prevented on the basis of the route. By comparing the impact of various levels of background concentration, it was found that when the background concentration was higher, the effect of route choice was smaller. However, if other pollutants and also short-term peaks in exposure are considered (which are commonly observed in active traveling), the number of preventable deaths could be significantly more.

It should be taken into account that active commuters are at high risk of air pollution exposure; they may inhale higher amounts of air pollution compared with other commuters, due to their close proximity to the source of air pollution, their
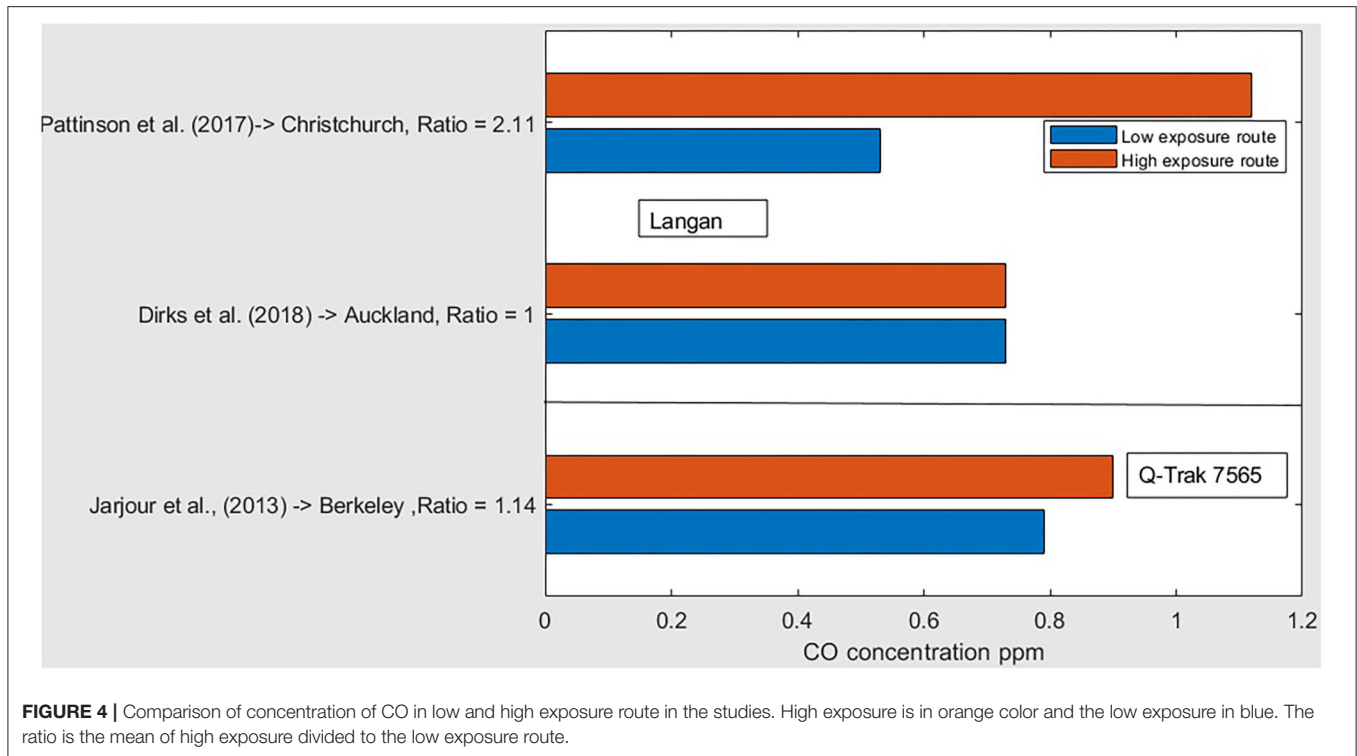
FIGURE 4 | Comparison of concentration of CO in low and high exposure route in the studies. High exposure is in orange color and the low exposure in blue. The ratio is the mean of high exposure divided to the low exposure route.

TABLE 2 | The reduction in exposure to pollutants by taking an alternative route in cycling (R = ratio of mean low exposure route to background, the unit for all numbers is $\mu g/m^3$).

	Mean high exposure	Mean low exposure	Difference	Background (R = 1)	Background (R = 2)	Background (R = 3)	Background (R = 4)	Background (R = 5)
Celje	6.20	4.90	1.30	4.90	2.45	1.63	1.23	0.98
Rotterdam	1.73	1.63	0.10	1.63	0.82	0.54	0.41	0.33
London	5.53	3.73	1.80	3.73	1.87	1.24	0.93	0.75
Sao Paulo	7.23	4.76	2.47	4.76	2.38	1.59	1.19	0.95
Antwerp	3.38	1.32	2.06	1.32	0.66	0.44	0.33	0.26
Berkeley	2.06	1.76	0. 3	1.76	0.88	0.59	0.44	0.35
Mean	4.83	3.61	1.21	3.61	1.81	1.20	0.90	0.72

TABLE 3 | Mean annual BC exposure by taking each route (L = low exposure, H = high exposure route, the unit for all numbers is $\mu g/h.m^3$, R = ratio of mean low exposure route to background).

	Route	Celje	Rotterdam	London	Sao Paulo	Antwerp	Berkeley	Mean
R = 1	L	4.900	1.630	3.730	4.760	1.320	1.760	3.613
	H	4.935	1.633	3.778	4.827	1.375	1.768	3.646
R = 2	L	2.516	0.837	1.915	2.444	0.678	0.904	1.855
	H	2.551	0.840	1.964	2.511	0.733	0.912	1.888
R = 3	L	1.721	0.573	1.310	1.672	0.464	0.618	1.269
	H	1.756	0.575	1.359	1.739	0.519	0.626	1.302
R = 4	L	1.324	0.440	1.008	1.286	0.357	0.476	0.976
	H	1.359	0.443	1.056	1.353	0.412	0.484	1.009
R = 5	L	1.086	0.361	0.826	1.055	0.292	0.390	0.800
	H	1.121	0.364	0.875	1.121	0.348	0.398	0.833

TABLE 4 | Estimated attributable proportion by taking the routes based on the measured black carbon (Diff = difference in attributable proportion, L = low exposure, H = high exposure route).

	Route	Celje	Diff (%)	Rotterdam	Diff (%)	London	Diff (%)	Sao Paulo	Diff (%)	Antwerp	Diff (%)	Berkeley	Diff (%)	Mean	Diff (%)
R = 1	L	24.84	0.15	9.06	0.01	19.53	0.23	24.22	0.29	7.40	0.29	9.75	0.04	18.98	0.15
	H	24.99		9.08		19.76		24.52		7.70		9.79		19.14	
R = 2	L	13.64	0.18	4.76	0.01	10.56	0.25	13.27	0.34	3.87	0.31	5.13	0.04	10.25	0.17
	H	13.81		4.77		10.81		13.61		4.18		5.17		10.42	
R = 3	L	9.54	0.18	3.28	0.02	7.35	0.26	9.28	0.35	2.67	0.31	3.54	0.05	7.13	0.18
	H	9.73		3.30		7.61		9.63		2.98		3.58		7.31	
R = 4	L	7.42	0.19	2.53	0.02	5.70	0.27	7.22	0.36	2.06	0.32	2.73	0.05	5.53	0.18
	H	7.61		2.55		5.97		7.58		2.37		2.78		5.71	
R = 5	L	6.13	0.19	2.08	0.02	4.70	0.27	5.96	0.36	1.69	0.32	2.25	0.05	4.56	0.18
	H	6.32		2.10		4.97		6.32		2.01		2.29		4.74	

often longer travel times and their increased minute ventilation (Bigazzi and Figliozzi, 2014; Cepeda et al., 2017; Targino et al., 2018). Additionally, exposure to spikes in pollution (a high concentration of air pollution for a short period of time) during walking or cycling may also result in adverse health effects (Michaels and Kleinman, 2000; Dons et al., 2019). However, it has been reported that as long as the average air pollution exposure throughout the commute remains below 1.1 and 2 times the background level for a cyclist and walker, respectively, on average, the benefits of active travel can be considered to outweigh the health risk of air pollution exposure (Johan de Hartog et al., 2010). However, exposure to traffic-related pollution may negate the benefits of active travel when commuting along high-trafficked routes, particularly for people with pre-existing conditions such as COPD and ischemic heart disease (Sinharay et al., 2018).

There are some limitations associated with this review that need to be recognized. Although the search terms used were comprehensive, the review was limited to articles written in English; non-English studies were excluded. The lack of Asian and African studies is acknowledged. A small number of cities in this study were analyzed. All of the cities except one (Sao Paulo) were from developed countries, indicating a lack of studies in least developed and developing countries. However, had a larger number of studies from cities from developing, least developed as well as developed countries been included in the study, a larger range of differences in exposure between low and high exposure routes may have been found. In this study, for each of the publications included in the review, only two routes were investigated: a high exposure route and a low exposure route. However, many active commuting routes include a mix of low and high exposure sections along the route, sometimes with choice available, and sometimes not. Thus, is may not always be possible to choose a low exposure route for the whole of the journey. In the present study, only the air pollution exposure along the routes was considered, with other factors influencing an active commuter's route choice, including the presence of trees and the route topography, ignored.

The Health Impact Assessment aspect of this study included a number of assumptions. For example, we assumed that the commuter spends 1 h per weekday traveling. Additionally, the estimates of health impacts are based only on what is known about the health effects associated with one pollutant (BC). Had other pollutants been considered, the results might have been different. However, BC is considered to be one of the pollutants most strongly associated with adverse health outcomes (Magalhaes et al., 2018). Additionally, only the long-term health impacts were considered; the short-term effects induced by exposure to spikes in air pollution, should also be taken into account in evaluating the risks associated with high exposure routes compared with low exposure routes.

It is also noted that there are limitations associated with the use of AirQ+ for estimating the health risks. For example, the estimate of relative risks rely on studies taken place in situations of high pollution (World Health Organization, 2020a). Also, the model equations are based on studies carried in Western Europe and North America (World Health Organization, 2020a), with little input provided from other continents with cities also affected by traffic-related air pollution. Therefore, model predictions may not be reliable for these regions. The model also assumes that ambient concentrations are reliable indicators of population exposure (World Health Organization, 2020a). Thus, invariably, there is a risk of, introducing bias in the estimation as a result (Evangelopoulos et al., 2020). Furthermore, the equations used for estimating the health risks are based on concentration-response functions in which many assumptions have been made. This includes issue of transferability, the generalizability of health impacts from one population to another, and the use of individual pollutants derived from single-pollutant statistical models rather than from a mixture of pollutants (World Health Organization, 2020a), which may or may not be realistic, depending on the specific situation.

This research has identified many questions that are also worthy of further investigation. In particular, future research is needed to examine more closely the links between the nature of vehicle fleets and traffic congestion, and their impact on air pollution exposure. Additionally, there is limited information on the health effects associated with exposure to short-term spikes in air pollution. Thus, there is abundant room for further progress to be made in terms of understanding the health consequences

of exposure to the intermittent spikes in concentration typical of traffic pollution concentration traces.

Furthermore, there remains a lack of research regarding children and elderly people, especially given these groups are more vulnerable to the adverse effects of air pollution (Friedrich, 2018). Children's organs are not fully developed, hence their respiratory and immune systems are less efficient than those of adults (Zielinska and Hamulka, 2019). Moreover, epidemiological studies suggest that excess mortality due to air pollution is greater among older age groups (especially 75+) (Simoni et al., 2015).

The review also suggests that there is a lack of studies focused specifically on walking. Walkers are more susceptible to traffic-related air pollution due to the long travel time as walking generally takes longer to travel the same distance compared to other modes of transport, and because of walkers' close proximity to traffic. Walkers are also at risk of exposure to short-term spikes since there are no physical barriers between the source of traffic pollution and pedestrians (Dirks et al., 2016). Therefore, more research is required to determine the benefits of choosing alternative routes with respect to exposure amongst children and the walking mode of transport.

It is also acknowledged that in many cities, particularly in developing or least developed countries, the selection of low exposure routes and the associated avoidance of exposure to air pollution might simply not be feasible. In the short term, an economic alternative could be to encourage active travelers to wear effective masks, such as N95, that can result in significant reductions in exposure to airborne particles (Kyung et al., 2020). This could be particularly effective for susceptible groups of the population and people using various open vehicles (e.g., motor scooters, open delivery vehicles, etc.) who are exposed to large amounts of traffic pollution due to their mode choice and the amount of time spent in the transport microenvironment. It is noted that wearing a mask should be considered to be a short-term solution to limit exposure to air pollution exposure in anticipation of more effective solutions, including the widespread introduction of clean vehicle technology.

CONCLUSIONS

The main goal of the current study was to review studies investigating the role of route choice on the air pollution exposure of active mode commuters. The results clearly indicate that significant reductions can be obtained by choosing low traffic routes away from traffic congestion. Moreover, identifying the

cleanest path from the point of view of air pollution is important in active commuters' route planning, as even a modest reduction in daily air pollution exposure can lead to a significant reduction in exposure to a population as a whole, or for an individual when considering their cumulative exposure over a period of years. People, especially those who walk/cycle the same route every day, would benefit from information about air pollution for the purpose of route planning.

Governments play a crucial role in helping to reduce active commuters' exposure to air pollution by the provision of transport infrastructure that is conducive to improved localized air quality. Although the most important factor in active commuters' route choice is often the distance involved or the number of directional changes required, route choice is a complex function of other parameters, considered in varying amounts, including safety aspects and route aesthetics. Planners should consider identifying and targeting paths of low exposure, in addition to active commuters' influencing factors regarding preferred routes when considering further infrastructure development, thus encouraging more journeys along such routes. Furthermore, active commuters themselves may also have an important role to play, with opportunities for community participation via the use of mobile apps to help in finding the least polluted routes and in raising awareness regarding air pollution (Mahajan et al., 2020). This would help to increase the levels of participation in active transport and makes the city more liveable (Koh and Wong, 2013) while promoting active commuters' health.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/supplementary materials, further inquiries can be directed to the corresponding author/s.

AUTHOR CONTRIBUTIONS

Data extraction was conducted by MR. It was subsequently sent to KD for cross-checking and consultation. MR screened all of the papers and carried out the health impact assessment. The extracted data were crosschecked by KD who monitored all levels of the screening. AW suggested adding a health impact assessment. KD supervised the project. All authors discussed the results and contributed to the final manuscript by providing critical feedback.

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