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Benefits of shift from car to active transport

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ABSTRACT

There is a growing awareness that significant benefits for our health and environment could be achieved by reducing our use of cars and shifting instead to active transport, i.e. walking and bicycling. The present article presents an estimate of the health impacts due to a shift from car to bicycling or walking, by evaluating four effects: the change in exposure to ambient air pollution for the individuals who change their transportation mode, their health benefit, the health benefit for the general population due to reduced pollution and the risk of accidents. We consider only mortality in detail, but at the end of the paper we also cite costs for other impacts, especially noise and congestion. For the dispersion of air pollution from cars we use results of the Transport phase of the ExternE project series and derive general results that can be applied in different regions. We calculate the health benefits of bicycling and walking based on the most recent review by the World Health Organization. For a driver who switches to bicycling for a commute of 5 km (one way) 5 days/week 46 weeks/yr the health benefit from the physical activity is worth about 1300 €/yr, and in a large city (> 500,000) the value of the associated reduction of air pollution is on the order of 30 €/yr. For the individual who makes the switch, the change in air pollution exposure and dose implies a loss of about 20 €/yr under our standard scenario but that is highly variable with details of the trajectories and could even have the opposite sign. The results for walking are similar. The increased accident risk for bicyclists is extremely dependent on the local context; data for Paris and Amsterdam imply that the loss due to fatal accidents is at least an order of magnitude smaller than the health benefit of the physical activity. An analysis of the uncertainties shows that the general conclusion about the order of magnitude of these effects is robust. The results can be used for cost-benefit analysis of programs or projects to increase active transport, provided one can estimate the number of individuals who make a mode shift.

1. Introduction

There is a growing awareness of the need to change our transportation habits by reducing our use of cars and shifting instead to active transport, i.e. walking and bicycling. Such change can bring about significant benefits for our health and environment. To help policy makers, urban planners and local administrators make the appropriate choices, it is necessary to quantify all the significant impacts of such a change. There are countless possible effects, some of which are extremely difficult to evaluate, for instance impacts on the social fabric of a community, on the sense of well-being of the population, even on the crime rate. But health impacts of the physical activity (PA) and of air pollution are especially important, and at least their associated benefit in terms of reduced mortality can be evaluated quite reliably.

Two recent studies have carried out such an assessment for specific cities or regions: Woodcock et al. (2009) evaluated the health impacts that can be expected for London and for New Delhi, and de Hartog et al. (2010) evaluated mortality impacts for the Netherlands. For the benefits of reduced air pollution these studies used detailed site-specific models for atmospheric dispersion and chemistry. Unfortunately it is not clear how such results can be transferred to other sites. Rojas-Rueda et al. (2011) evaluated the health benefit of the bike sharing program in Barcelona; they included the effect of pollution exposure for the bicyclists, but not the public benefit due to reduced vehicle emissions.

In the present paper we carry out a similar assessment of the health impacts, but to calculate the population exposure to air pollution we use results of the most comprehensive assessment of automotive pollution impacts in Europe, namely the transportation study of ExternE (2000) (ExternE, "External Costs of Energy", is a multidisciplinary and multinational project series of the European Commission DG Research that has been continuing since 1991). This allows us to derive generic estimates that can

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be applied to a wide range of sites: large cities, small cities and rural areas, even outside the EU. By contrast to the limitations of a site-specific study we offer our analysis in the spirit of "better approximately right than precisely wrong". In addition to our detailed analysis of PA and air pollution we also look at accident statistics, and we cite external cost estimates for further benefits of active transport: reduced CO_2 emissions, noise and congestion. We include a wider range of impacts than Woodcock et al. (2009) and de Hartog et al. (2010), and for the health benefits of active transport we use the most recent reviews by the World Health Organization (WHO, 2008, 2010).

We calculate results per individual driver who switches to active transport. We consider a trajectory of 5 km for bicycling (and 2.5 km for walking) and provide a detailed evaluation of four effects when people change their transportation mode from driving to bicycling or walking:

- the health benefit of the physical activity,
- the health benefit for the general population due to reduced pollution,
- the change in air pollution impacts for the individuals who make the change,
- and changes in accidents.

There is a wide variety of possible health impacts, but here we focus on mortality, because the dose-response functions and accident data for this end point have the lowest uncertainty. In monetary terms the mortality impacts are especially large, and they also tend to weigh heavily in public perception. But we also indicate how the conclusions might change if other health endpoints are included.

The inclusion of other endpoints and of items such as congestion implies a variety of incommensurate impacts that would complicate any practical application of the results, unless one uses monetary valuation to measure all the impacts on a common scale. For that reason we present our results in monetary terms, while noting that simple division of the mortality costs by the respective unit costs yields the corresponding changes in life expectancy and number of deaths.

Our calculations require only a simple spreadsheet and we document all the equations and parameters, to enable the reader to modify the parameter choices and see the consequences. We also analyze the uncertainties.

We have tried to provide estimates for all the effects that appear to be most important in monetary terms, both for the individuals who switch their transport mode and for the general public. The results can be used for cost-benefit analysis of programs and projects that encourage active transport, if one can estimate the number of individuals who are induced to switch their transport mode. But that number may be very difficult to determine, as we find when we attempt a comparison of costs and benefits of a large and politically important bike sharing program, the Vélib program of Paris.

2. Concepts, tools and literature

In this section we describe the general concepts and tools, before proceeding to detailed implementation in Section 3. To begin we list abbreviations and acronyms in Table 1.

Table 1
Abbreviations and acronyms.

CI	Confidence interval
COPERT	Software to determine vehicle emissions
DRF	Dose-response function
EU	European Union (added number indicates number of member states included)
ExternE	External Costs of Energy = project series of EU to determine external costs
LE	Life expectancy
MET	Unit for measuring metabolic rates
PA	Physical activity
PM	Particulate matter
$PM_{2.5}$	Particulate matter with diameter less than 2.5 μm
RR	Relative risk
s_{DR}	Slope of dose-response function
VOLY	Value of a life year
VPF	Value of prevented fatality
WHO	World Health Organization

2.1. Monetary valuation

As explained in the introduction, we use monetary valuation to present a wide variety of incommensurate impacts on a common scale. For the monetary valuation of fatal accidents we take a value of a prevented fatality[1] (VPF) of 1,600,000€$_{2010}$, typical of what is used for traffic accidents in the EU.[2] For PA and air pollution, by contrast, we base the valuation of mortality on the change in life expectancy (LE), taking the value of a life year (VOLY) equal to 40,000€$_{2006}$, according to a contingent valuation study in nine countries of the EU (Desaigues et al., 2011) which has been adopted by ExternE. The main reason for choosing a different valuation for accidents lies in the nature of the deaths: on average a traffic fatality causes the loss of about half a life span, on the order of 40 yr, whereas most air pollution deaths occur among individuals who are very frail because of old age or poor health and their LE loss is relatively short: for typical exposures in Europe and North America the population-average LE loss due to pollution is only about eight months. Furthermore, as shown by Rabl (2003), the total number of deaths attributable to air pollution cannot even be determined, whereas the LE loss can be calculated unambiguously from the relative risk (RR) numbers of epidemiological studies of chronic air pollution (see the review by Chen et al. (2008)), using standard life table methods. Likewise the LE gain from PA is relatively short, around 1 yr for our bicycling scenario, and a valuation is more appropriate in terms of VOLY than VPF. Correcting for inflation we take VOLY equal to 43,801€$_{2010}$.

2.2. Benefits of physical activity

That physical activity brings large health benefits has been established beyond any doubt, by countless epidemiological studies in many countries all over the world, as shown for example in the review by the US Department of Health and Human Services (US DHHS, 2008). We use this review, which presents explicit dose-response functions (DRF) for several end points, as a basis for our calculations because it is the most comprehensive we have found. In particular we use the DRF for all-cause mortality, shown here as a solid line in Fig. 1, drawn as a linear interpolation of the data points (the other lines in this figure will be explained in Section 3). The data points represent the median of the DRFs of 12 studies that are sufficiently comparable to be summarized in such manner. The general pattern is typical of the various health benefits of PA; it is nonlinear, the incremental benefit being greatest at low levels of activity.

[1] Economists have usually called this quantity "value of statistical life", a most unfortunate term that tends to evoke hostile reactions among non-economists. It is not the intrinsic value of life but the willingness to pay to avoid an anonymous premature death, and VPF is a better term.

[2] In the USA much higher values are used, around \$6 million.

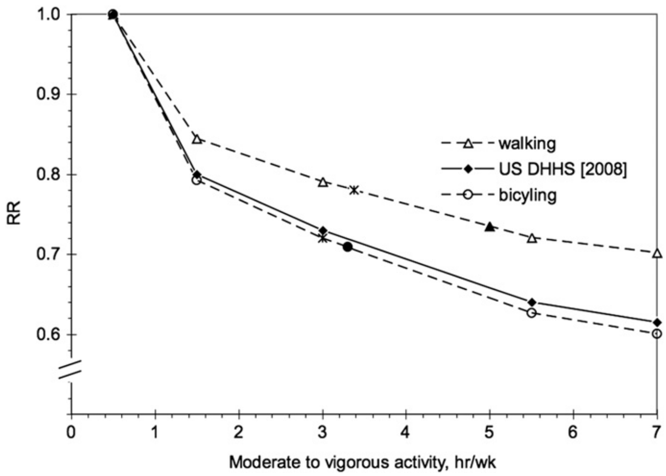

Fig. 1. DRF for relative risk of all-cause mortality, as function of hours/week of physical activity. Solid line: data of US DHHS (2008). Dashed lines are obtained by scaling (1 − RR) in proportion to the (1 − RR) of WHO (2010) for walking and of Andersen et al. (2000) for bicycling at the points indicated by the stars. The black points on the dashed lines indicate the RRs chosen for our scenarios.

In addition to mortality, PA also reduces the incidence of a wide range of morbidity endpoints, especially coronary heart disease, stroke, hypertension, and type 2 diabetes; PA is also associated with significantly lower rates of colon and breast cancer, as well as improved mental health (US DHHS, 2008). The range of morbidity benefits is much wider than for air pollution where morbidity involves mostly cardio-pulmonary effects. In monetary terms the ratio of morbidity over mortality benefits may thus be significantly larger than the ratio 0.5 that ExternE finds for air pollution, but further research is needed to examine this question.

For the health benefits of bicycling we invoke WHO (2008). The authors of this report carried out a thorough review of health benefits of bicycling and concluded that it would be best to consider only mortality, using as basis a large epidemiological study of cyclists in Copenhagen (Andersen et al., 2000). They also developed a software package, called HEAT, that calculates the mortality benefits of bicycling. Here we do not use HEAT because it evaluates mortality in terms of deaths rather than life expectancy change.

The study by Andersen et al is a prospective cohort study of the effects of PA on all-cause mortality, involving 30,896 men and women, with mean follow-up of 14.5 yr. The bicycling results are based on the subset of 6954 individuals who bicycle to work. Such large sample and follow-up was possible because Copenhagen is one of the cities with the highest percentage of bicycling to work, more than 35%. After adjustment for age, sex, educational level, leisure time physical activity, body mass index, blood lipid levels, smoking, and blood pressure, the relative risk was RR = 0.72 (95% CI, 0.57–0.91) for individuals who bicycle to work (average 3 h/week) compared to those who do not. The individual variability of the benefit, due to the nonlinearity of the DRF, is implicitly taken into account by virtue of averaging over all individuals in the age group.

The World Health Organization is in the process of extending the HEAT software to include walking. Even if the software tool is not yet ready, the key parameter for the estimation of the mortality reduction has been chosen, based on a review and meta-analysis of nine studies (WHO, 2010). The recommended relative risk for the reduction of mortality is RR = 0.78 (95% CI: 0.64–0.98) for a walking exposure of 29 min seven days a week = 3.38 h/week.

2.3. Car emissions

To estimate the emissions of a car, we use the COPERT4 software, version 8.0, of the European Environment Agency [downloaded 4 Jan. 2011 at http://lat.eng.auth.gr/copert/]. The user specifies the vehicle types, as well as the percentage of each of three main driving conditions (urban, rural and highway) and the corresponding average speed. Vehicle types are specified in terms of EURO standards, for gasoline or diesel, respectively; they apply to new cars sold after the respective enforcement dates. We consider passenger cars conforming with the EURO4 and EURO5 standards, under conditions of urban driving. EURO4 has been in force since January 2005, and EURO5 is fully in force since January 2011.

Ideally one should take life cycle emissions rather than just the tail pipe emissions of COPERT4. Life cycle emissions can be estimated by means of the GREET software for Well-to-Wheel analysis (ANL, 2004). However, for vehicles with conventional fuels the upstream emissions are relatively small, on the order of 25%, and they occur in regions with relatively low population density. Since the health effects of concern are due to local impacts of $PM_{2.5}$ emissions in cities, as explained in Section 2.4, the contribution of upstream $PM_{2.5}$ emissions is entirely negligible.

2.4. Health impacts of air pollution

The health impacts of air pollution have been the focus of intense research worldwide and the results have been used for health impact assessment and calculation of external costs by organizations such as WHO (2003), EPA (Abt, 2004), NRC (2009) and the EC (ExternE, 2000, 2005; CAFE, 2005). The assumptions made by these studies are quite similar. Here we use the methodology and results of ExternE for air pollution, both for the dose-response functions (DRF) and for the estimation of the population exposure. As far as mortality is concerned, a correct assessment of the total mortality impact requires DRFs for chronic exposure (Rabl, 2006), rather than DRFs determined by time series studies because the latter take into account only acute effects of short term exposure.

The standard approach taken by almost all studies that have quantified the health impacts of air pollution, in particular ExternE, EPA and WHO, is to use only DRFs for PM and for O_3. Direct effects of NO_x and SO_2 are assumed to be negligible but the secondary nitrate and sulfate aerosols created by their transformation in the atmosphere are considered as PM and their impacts are calculated by using the DRFs for PM. The reasons for this choice are that the DRFs for PM and O_3 are better established than for NO_x and SO_2, and that pathways of action within the body have been identified for primary combustion particles and for O_3 whereas it is less clear how NO_x or SO_2 could have harmful effects at the low concentrations typically found in the ambient air. As for the size specification of PM, there is an emerging consensus that $PM_{2.5}$ is more relevant than PM_{10}. Even though there are questions about the toxicity of nitrate and sulfate aerosols (Reiss et al., 2007), the standard approach yields correct results for assessments of the total health impact of typical urban ambient concentrations because it uses DRFs that are based on typical urban ambient PM with its mix of primary and secondary particles. Thus this approach is appropriate for evaluating the effects of exposure changes for the individuals who make a mode switch (item 5 in Table 2) if one uses, as we do, measured ambient PM data.

For the public benefit of reduced emissions (item 4 in Table 2), however, we have to evaluate something quite different, namely the contribution of a specific incremental pollution source rather than the effect of ambient concentrations (which are due to a variety of sources as well as chemical reactions in the atmosphere). For the impacts of primary pollutants emitted at ground

Table 2
Key assumptions.

(1) Scenarios
a) Use bicycle instead of car for commuting to work 5 days/week, 46 weeks/yr trajectory 5 km one way, 2300 km/yr, by car: average speed 20 km/h, duration of one-way trip 0.25 h, by bicycle: average speed 17 km/h, duration of one-way trip 0.33 h.
b) Walk instead of driving for commuting to work 5 days/week, 46 weeks/yr trajectory 2.5 km one way, 1150 km/yr, by car: average speed 20 km/h, duration of one-way trip 0.125 h, on foot: average speed 5 km/h, duration of one-way trip 0.5 h.
(2) Benefit of PA
Life table calculation of LE change, with the following RR
a) for bicycling: based on Andersen et al. (2000) and applying a correction for the difference of bicycling duration compared to our scenario, assume RR=0.709 for age-specific mortality from age 25 to age 65, as result of bicycling from age 20 to age 60,
b) for walking: based on WHO (2010) and applying corrections for our scenario, assume RR=0.735 for age-specific mortality from age 25 to age 65, as result of walking from age 20 to age 60.
(3) Health impacts of air pollution
DRF for mortality due to $PM_{2.5}$ is linear without threshold and is expressed as LE loss, with slope s_{DR}=6.50E−04 years of life lost per person per year per $\mu g/m^3$ of $PM_{2.5}$, based on Pope et al. (2002) and ExternE (2005). Impact change of individuals is proportional to duration of exposure/dose change.
(4) Public benefit from reduced pollution
a) Avoided emissions: 0.031 $g_{PM2.5}$/km, based on COPERT 4 software.
b) Calculation of avoided air pollution mortality: based on results of the Transport phase of ExternE (2000), but updated to current best values for DRF and monetary valuation.
(5) Effect of exposure change from car to bicycle and from car to walking
Based on measured concentration data in representative busy streets of eight cities of EU (EEA, 2008), assume 23 $\mu g/m^3$ of $PM_{2.5}$ and 57 $\mu g/m^3$ of NO_2 at side of street.
Modifying factors for exposure (due to increased concentration) and dose (due to increased inhalation) during different transport modes: 1.5 for cars, 2 for pedestrians, 3 for bicyclists.
(6) Accidents
Accident statistics for Paris, Belgium and the Netherlands.
Cost of nonfatal bicycle accidents based on Belgian data of Aertsens et al. (2010).
(7) Monetary valuation
Monetary valuation of fatal accidents based on VPF=1.6M€$_{2010}$.
Monetary valuation of PA and air pollution based on VOLY=43,801€$_{2010}$
Cost of CO_2 emissions based on 25 €$_{2010}$/tonne$_{CO2}$

level in large cities the regional contribution is negligible compared to the local contribution, as explained in Section 2.6 below (for details, see Table 4 in Section 3.6). Since the formation of nitrate and sulfate aerosols is slow and takes place over distances of tens to hundreds of km, their local contribution is negligible. The local contribution of O_3 is also negligible because it is a secondary pollutant created gradually in a region of tens of km from the source, and in the city the concentration is actually reduced by cars because much or most of their NO_x emission is in the form of NO which destroys O_3 locally, before causing the creation of O_3 further away.

Thus the standard approach limits our analysis to primary pollutants and specifically to $PM_{2.5}$, while totally neglecting NO_x, the other pollutant emitted in large quantities by cars. This despite the fact that many experts consider NO_2 a valid indicator for the severity of automotive pollution, and there are numerous epidemiological studies that have found significant associations, but only for acute NO_2 exposure. In their meta-analysis of effects of chronic exposure Chen et al find nothing significant for NO_2: their RR_{10} for all-cause mortality is 1.0 (95% CI: 0.99–1.02), RR_{10} being for a 10 $\mu g/m^3$ increment. For other end points they do find positive associations for NO_2 but none are statistically significant: RR_{10}=1.04 (95% CI: 0.96–1.12) for any cardiovascular event (incidence and mortality), RR_{10}=1.11 (95% CI: 0.99–1.24) for incidence of lung cancer and RR_{10}=1.01 (95% CI: 0.94–1.09) for mortality from lung cancer. The heterogeneity between the respective studies is large, reflecting the difficulties of determining the exposure (the variability of individual exposure relative to concentrations observed by measuring stations is much larger for NO_2 than for PM). If one were to include DRFs for NO_2, it would not be clear to what extent the effect should be added to those of $PM_{2.5}$, if NO_2 is merely an indicator of pollution and not the causative constituent. There are also various additional automotive pollutants, e.g. aliphatic hydrocarbons, benzene, butadiene, and formaldehyde, but their quantities and/or DRF slopes are so low that their health impacts are negligible compared to $PM_{2.5}$. In view of this situation we follow the standard approach and consider only $PM_{2.5}$.

2.5. *Change in exposure for individuals who switch from car to bicycle or to walking*

Several studies have measured the exposures of drivers and bicyclists on selected trajectories, for example AIRPARIF (2009) in Paris, ORAMIP (2008) in Toulouse (France), Zuurbier et al. (2010) in Arnhem (The Netherlands) and Int Panis et al. (2010) in Brussels, Louvain-la-Neuve and Mol (Belgium). The data show that the change in exposure of individuals who leave their car to bicycle or to walk is extremely variable from one case to another. However, as our calculations will show, this does not matter since the health impact of such changes is entirely negligible compared to the overall benefits of the physical activity.

As a starting point we take the concentrations that have been measured in streets of large cities. For European cities such data have been reported in Fig. 5.2 of EEA (2008). This figure shows annual average concentrations for monitoring stations along busy roads in major European cities: Vienna, Prague, Paris, Berlin, Athens, Krakow, Bratislava, Stockholm and London for NO_2, and Prague, Copenhagen, Berlin, Reykjavik, Rome, Bratislava, Stockholm and London for PM_{10}. Numbers for NO_2 are shown for each of the years 1999 to 2005; they vary slightly around 57 $\mu g/m^3$, without any clear long term trend and significantly above the 40 $\mu g/m^3$ specified as upper limit by the air quality guidelines of the WHO (2005). Unfortunately the EEA report has no data for $PM_{2.5}$. Numbers for PM_{10} are shown for each of the years 2002 to 2005; they vary between 40 and 37 $\mu g/m^3$, with a slight declining trend. To estimate the corresponding values for $PM_{2.5}$, we multiply 38 $\mu g/m^3$ by a typical ratio of $PM_{2.5}/PM_{10}$=0.6 to obtain 23 $\mu g/m^3$. This, too, is well above the WHO guideline of 10 $\mu g/m^3$.

The exposures encountered by the commuters depend on the detailed conditions of each trip. Concentrations inside a car tend to be higher than roadside concentrations, but in newer cars with good air filters the exposure can be much lower. A cyclist in the middle of a busy street is exposed to concentrations higher than the side of the road, but on a separate bike path the exposure could be up to two times lower. Here we assume that the concentrations of $PM_{2.5}$ and NO_2 inside a car are 50% higher than the roadside concentrations measured by EEA whereas the bicyclist is exposed to the roadside concentration. We also take the roadside concentration for pedestrians.

Whatever the exposure, one also has to account for the fact that the pollutant dose increases with the inhalation rate. Both the number of breaths per minute and the volume per breath increase (Int Panis et al., 2010). Here we assume that the dose is proportional to the total air intake, and that the latter is proportional to the metabolic rate. This assumption agrees with detailed calculations (de Nazelle et al., 2009), using the algorithms of Johnson (2002), within about 25% in the MET range of interest, an approximation that is certainly adequate in view of the much larger uncertainties of the real exposures and of typical metabolic rates for our scenarios. Metabolic rates are expressed as Metabolic Equivalent (MET), one MET being defined as 1 kcal/kg/h, which is roughly equal to the energy cost of sitting quietly. Metabolic rates for different activities have been measured systematically, see e.g. Ainsworth et al. (2000). A detailed catalog of MET values (http://prevention.sph.sc.edu/tools/docs/documents_compendium.pdf) shows the following:

Rest	1.0 MET
Transportation: riding a car or truck	1.0 MET
Transportation: automobile or light truck driving	2.0 MET
Walking: 2.5 mph (miles/h), firm surface	3.0 MET
Walking: 2.0 mph, level, slow pace, firm surface	2.5 MET
Bicycling: <10 mph, leisure, to work	4.0 MET
Bicycling: 10–11.9 mph, leisure, slow, light effort	6.0 MET

2.6. Impact on the general public

To estimate the mortality impact for the general population, we use results of ExternE (2000) because it is still the most comprehensive assessment of the impacts of vehicle emissions in the EU. The concentrations due to vehicle emissions were calculated with the RoadPol Gaussian plume model (Vossiniotis et al., 1996) in the local zone (up to about 25 km of the source). Beyond the local zone a Lagrangian trajectory model with chemical reactions was used, covering the entire European continent. However, for primary pollutants emitted at ground level in large cities around 95% of the impact is within the local zone; the local contribution of secondary pollutants is negligible because they are created far from the source. These atmospheric models are combined with population data, DRFs and monetary values in the EcoSense software of ExternE.

The impact of primary pollutants emitted at ground level in large cities depends strongly on the detailed relationship between the site where the emission takes place and the distribution of the population. Nonetheless the results of ExternE (2000) indicate that one can draw approximate general conclusions, as we will discuss in Section 3.6.

2.7. Accidents

Changes in accidents are difficult to estimate, because they are extremely dependent on the specifics of the change: even though bicyclists are more vulnerable than drivers, their accident risk can become very small or negligible if bike paths are provided or if bicycling is as widely adopted as in the Netherlands or Denmark (in Amsterdam and Copenhagen more than a third of the commuters use the bicycle). Quite generally nationwide fatality rates per km are higher for bicyclists than for cars. However, one must be careful in interpreting the statistics. In particular, the rates per km are very different between rural and urban areas, both for cars and for bicycles. A major difficulty in estimating the rates of fatal bicycle accidents lies in the fact that they are rare events.

There is enormous variability between different countries and cities, the rates being much lower in countries such as the Netherlands and Denmark where bicycling is widely practiced, because in such countries traffic management is better adapted to bicycling and both drivers and bicyclists have learned to coexist—there is safety in numbers. This phenomenon can be seen very clearly in Figs. 1 and 2 of Vandenbulcke et al. (2009) where the bicycling rates and accident rates for different regions of Belgium are shown: accident rates (in terms of serious accidents per minute of bicycling) are roughly an order of magnitude lower in areas where the bicycle use for commuting is high (12.8–21.7%, in the North of Belgium) than in areas where such bicycle use is low (less than 2.2%, in the south of Belgium). Pucher and Buehler (2008) show that fatality rates per 100 million km bicycled range from 1.1 in the Netherlands to 3.5 in Italy in the EU; in the USA the rate is 5.8. For pedestrians Pucher and Dijkstra (2000) show that fatality rates per km traveled in Germany and the Netherlands are approximately the same as for bicycles.

One should account for all the avoided deaths due to car accidents when people switch from car to active transport. Whereas the probability of a driver getting killed during a commute in a large city is small, one also has to consider pedestrians and bicyclists killed by cars. Unfortunately it is difficult to get reliable statistics. de Hartog et al. (2010) cite a study for the Netherlands (Dekoster and Schollaert, 1999) that compared the risks of a fatal accident for car drivers and cyclists, including the risk to other road users: considering only roads used by cars and by bicycles, they find that the total number of fatalities per km traveled is essentially the same for cars and for bicycles. That is unlikely to hold for countries where bicycling is less common than in the Netherlands, as we show in Section 3.7 with explicit data for France.

3. Specific assumptions

3.1. Summary of key assumptions

We begin by choosing the scenarios, namely a change in the transport mode for commuting to and from work. For the assessment of bicycling we consider an individual who switches from car to bicycle for a trajectory of 5 km one way. The assumptions for trip duration and average speed are typical of bicycling. For cars they are realistic for typical congestion in large cities; for smaller cities or rural sites the speed would be higher and the emission of pollutants per km somewhat lower. For a switch from car to walking the typical distance would be much shorter, commuting time being a crucial determinant for the choice of transportation mode; here we assume 2.5 km one way.

Table 2 indicates key assumptions and references. The following subsections present more detail.

3.2. Benefits of physical activity

Our scenario involves a bicycling time of 3.3 h/week, different from the 3 h/week of Andersen et al. Since the DRF is a nonlinear function of both level and duration of the physical activity, we adjust the RR of Andersen et al by assuming that the variation with duration follows the shape of the DRF of US DHHS (2008) (solid line in Fig. 1). Specifically, we derive a DRF for bicycling by assuming that the risk reduction $(1-\mathrm{RR})$ for bicycling is proportional to $(1-\mathrm{RR})$ of US DHHS (2008), the constant of proportionality being the ratio $(1-\mathrm{RR})_{\text{Andersen et al.}}/(1-\mathrm{RR})_{\text{US DHHS (2008)}}=0.28/0.27$ at the duration of 3 h/week indicated by the star in Fig. 1. This DRF is shown by the lower dashed line in Fig. 1. Reading this curve at 3.3 h/week we find the RR$=$0.709 for our bicycling scenario as indicated by the solid circle. For the confidence intervals we multiply the dashed curve by the ratios $(1-\mathrm{RR}_-)/(1-\mathrm{RR})=(1-0.57)/(1-0.72)$ and $(1-\mathrm{RR}_+)/(1-\mathrm{RR})=(1-0.91)/(1-0.72)$ of the lower and upper confidence intervals

RR_- and RR_+ of Andersen et al. In this way we find that $(1-RR)$ is 0.291, with confidence interval (0.094–0.447).

To derive the DRF for walking we use the same method as for bicycling, the constant of proportionality now being the ratio $(1-RR)_{\text{WHO (2010)}}/(1-RR)_{\text{US DHHS (2008)}}=0.22/0.284$ at the duration of 3.38 h/week indicated by the star. The resulting DRF for walking is shown by the upper dashed line in Fig. 1 and the RR for our walking scenario is 0.735 as indicated by the solid triangle. We find that $(1-RR)$ is 0.265, with confidence interval (0.024–0.434).

Like HEAT we consider a bicycling cohort of age 20–60 yr and assume a time delay of 5 yr for the full attainment of the benefit. Thus we assume that the age-specific mortality is reduced by a factor of 0.709 from age 25 to 65. We carried out life table calculations, using data for age-specific mortality for a wide range of countries, in particular for the EU in 2007 from Eurostat [http://epp.eurostat.ec.europa.eu/portal/page/portal/eurostat/home]. Since the Eurostat data cover only ages below 86 yr, we extrapolate to 108 yr by fitting the Gompertz formula to the mortality from age 40 to 85. The LE gain is 1.20 yr for EU25. It is not very different within the EU, varying by less than about 0.1 yr. For the USA the gain is 1.32 yr with data of 2006. The gains tend to be larger in countries with lower LE because lower LE is due to higher age-specific mortality, generally at all ages; thus a reduction of RR between 25 and 65 has a larger effect. In Romania where LE is only 73 yr, the LE gain from bicycling is 1.69 yr and for Russia the corresponding numbers are LE=67.5 yr and LE gain=2.67 yr.

Since these LE gains are the result of bicycling or walking from age 20 to 60, but we want an equivalent annual benefit, we multiply the LE gain by VOLY and divide by the 40 yrs from age 20 to 60. Such allocation per year, without discounting, is appropriate because discounting is already implicit in the VOLY of Desaigues et al. (2011). Multiplying the LE gain of 1.20 yr by VOLY we find that the average annual benefit of our bicycling scenario in the EU25 is 1310€ per year of bicycling. Similarly and assuming RR=0.735 for our walking scenario we find that the average LE gain in the EU25 is 1.09 yr, worth 1192€ per year of walking.

3.3. *Car emissions*

As explained above in Section 2.4, we assume that health impacts of car emissions are due only to $PM_{2.5}$. The COPERT results for car emissions are shown in Table 3. COPERT distinguishes between different cylinder sizes, but we show only simple averages over the respective cylinder sizes because the $PM_{2.5}$ emissions per km are the same while the CO_2 emissions (which increase somewhat with cylinder size) are not the main focus of our paper. We assume a rather low speed of 20 km/h because of congestion in large cities; for instance the measured average speed in Paris is approximately 20 km/h (EQT, 2004). Since a 50% gasoline 50% diesel mix of EURO4 is fairly representative of the current situation in the EU, we take 0.031 g/km $PM_{2.5}$ and 278.3 g/km CO_2 for the calculations in this paper. For higher speeds or cars of more recent vintage the emissions would be lower, and the reader could readily scale the public health impact in proportion to the emissions of Table 3, but in any case the $PM_{2.5}$ emissions per km do not vary much with speed. Rural emissions are lower, but we do not bother to indicate them because their public health impact is so small as to be negligible, as shown at the end of Section 3.6 below.

3.4. *Dose-response function for air pollution mortality*

Following ExternE we assume that the DRF for mortality due to chronic $PM_{2.5}$ exposure is linear without threshold and with

Table 3
Passenger car emissions for urban driving, as calculated by COPERT4. CO_2 is same for EURO4 and EURO5. Values in bold face are chosen for this paper.

g/km	**CO_2** at 20 km/h	**CO_2** at 50 km/h	**$PM_{2.5}$**, EURO4 at 20 km/h	**$PM_{2.5}$**, EURO4 at 50 km/h	**$PM_{2.5}$**, EURO5 at 20 km/h
Gasoline cars	306.7	198.7	0.012	0.011	0.012
Diesel cars	250.0	177.0	0.050	0.039	0.013
50% gas+50% diesel	**278.3**	187.8	**0.031**	0.025	0.013

slope[3] :

$$s_{DR}=6.50\text{E}-04 \text{ years of life lost per person per year per } \mu g/m^3 \text{ of } PM_{2.5}. \quad (1)$$

This DRF has been derived by means of a life table calculation of LE, assuming a relative risk of RR=1.05 for a 10 $\mu g/m^3$ increment of $PM_{2.5}$. That RR is the mean of the two estimates for all-cause mortality in the paper of Pope et al. (2002), and it is very close to the RR of 1.06 for the same increment obtained by Chen et al. (2008) in their meta-analysis.

3.5. *Change in exposure for individuals who switch from car to bicycle or to walking*

To determine the modifying factor for the DRF we assume that the MET rate for driving is the same as the 24 h population average that is implicit in the epidemiological studies of air pollution mortality. Based on all of the considerations in Section 2.5 we choose the following modifying factors to account for exposure (due to increased concentration) and dose (due to increased inhalation) during different transport modes. For cars we assume that the concentrations are 50% higher than what is reported by the measuring stations of EEA (2008) because the latter are at curb sides and at about 2 m above street level, whereas drivers in busy streets are much closer to the exhaust of other cars. Such levels have been observed by measurements in cars by e.g. AIRPARIF (2009). For pedestrians we assume the curb side data of EEA, together with a MET rate that is about twice the 24 h population average. For bicyclists we assume the curb side data of EEA, together with a MET rate that is about three times the 24 h population average. Thus our modifying factors are: 1.5 for cars, 2 for pedestrians, and 3 for bicyclists. For the change of the health impact we assume proportionality with the exposure duration. This choice of modifying factors is somewhat arbitrary, but for any reasonable choice the effect turns out to be negligible compared to the health benefit of the physical activity.

3.6. *Impact on the general public*

The impacts and external costs of vehicle emissions have been calculated by the Transport phase of ExternE in 2000. Specifically we refer to Section 13.8, p. 201–206 of ExternE (2000), Table 13.26 of which shows results for the damage cost of $PM_{2.5}$ emitted by cars in seven countries of the EU. In that study two emission sites were chosen in each country, one rural, the other a large city. Even though the selection of sites in that study did not follow any systematic criteria (some of the rural sites are much less urban than others), the results provide a fairly good indication of typical

[3] Eq. (6.11) of ExternE (2005), multiplied by a factor 1.625 for the conversion from PM_{10} to $PM_{2.5}$.

Table 4
Results for the damage cost in €$_{2000}$/kg (columns 2–4), of $PM_{2.5}$ emitted by cars in 7 countries of the EU, as calculated by ExternE (2000). The last column shows the cost of mortality in large cities, obtained by multiplying column 4 by the adjustment factor of Eq. (2).

Site[a]	Local €$_{2000}$/kg	Regional[b] €$_{2000}$/kg	Total €$_{2000}$/kg	Local/ total	Mortality €$_{2010}$/kg
Brussels (1.0, 1.8)	388.5	30.1	418.6	0.93	335.6
Helsinki (0.6, 1.3)	170.5	4.3	174.8	0.98	140.1
Paris (2.2, 11.8)			1170.0		938.0
Stuttgart (0.6, 5.3)	193.0	29.7	222.7	0.87	178.5
Athens (0.7, 3.1)	916.8	10.0	926.8	0.99	743.0
Amsterdam (1.4, 6.7)	361.9	22.1	384.0	0.94	307.8
London (7.6, 13)	675.0	30.1	705.1	0.96	565.3
Average					**458.3**

[a] The numbers next to the city name indicate the population in million, of the city and of the metropolitan area, mostly based on Wikipedia (the definitions of city and metropolitan area are not uniform).

[b] in Table 13.26 of ExternE (2000) the sum of Local and Regional is slightly larger than Total because of overlap of population grids; since the numbers for Total are correct, we have slightly reduced the ones for Regional to eliminate this overlap.

impacts in urban and rural areas. Here we show only the results for cities, reproduced in columns two to four of Table 4.

The local zone extends to about 25 km around the city, and the ratios in column 5 show that in large cities more than 90% of the total impact of $PM_{2.5}$ occurs in the local zone. The numbers in columns two to four include all health endpoints. The mortality cost was calculated with a DRF of 2.61E−04 years of life lost per person per year per μg/m^3 of $PM_{2.5}$ and a VOLY of 96,500€$_{2000}$; it is responsible for 71% of the total cost. For the present paper we use only mortality costs, hence we adjust for the mortality contribution of ExternE (2000) according to the current DRF, Eq. (1), and monetary valuation. Thus the entries in the last two columns are obtained by multiplying Total of column four by a factor

$$\text{Adjustment factor} = 0.71 \times \frac{6.50E{-}04\,\text{lifeyears}/(\text{person}\,\text{yr}\,\mu\text{g}/\text{m}^3)}{2.61E{-}04\,\text{lifeyears}/(\text{person}\,\text{yr}\,\mu\text{g}/\text{m}^3)} \times \frac{43{,}801€_{2010}}{96{,}500€_{2000}}. \tag{2}$$

In the following we take the mean for large cities, 458.3 €/kg of $PM_{2.5}$. For the rural data of Table 13.26 (not shown here) we find a mean of 28.2 €/kg of $PM_{2.5}$. In view of the result that even for emissions in large cities the public health benefit of active transport is small compared to the benefit of the physical activity, it is clear that for rural trips the public health benefit can be neglected.

3.7. Fatal accidents

Here we consider data for Paris and for Amsterdam, two cities that are very different in terms of bicycling. In Paris the number of bicycle trips (one way) is about 160,000 per day during weekdays, and the number of fatal accidents has been 5.3 per year between 2007 and 2009 (F. Prochasson, Préfecture de Paris, personal communication). This implies a rate of 6.6E−05 fatal accidents/yr per bicyclist, and with a valuation of 1.6 million €/death the cost is 105 €/yr per bicyclist. In Amsterdam there are about 7 bicycle deaths per year (Buehler and Pucher, 2010), but the number of bicycle trips is much higher, on the order of 570,000, implying a rate of 2.5E−05 fatal accidents/yr per bicyclist, with a cost of 39 €/yr per bicyclist.

We should also account for the avoided deaths (drivers, passengers and victims outside the car) from car accidents in cities when people stop driving, but it is difficult to obtain reliable data because most statistics are not sufficiently detailed. For the Netherlands de Hartog et al. (2010) argue, on the basis of a study by Dekoster and Schollaert (1999), that the total deaths per km are nearly the same for bicycles and for cars. In that case the net increase in fatalities due to a shift from car to bicycle is essentially zero for our scenario. That may well be the case for the Netherlands where drivers and bicyclists have learned to coexist.

But it is not the case for France. Here the official traffic accident statistics (ONISR, 2009) provide data for accidents in cities, on p.302, indicating the number of drivers and passengers killed for each vehicle type in 2009 (for car accidents it is 216 drivers and 98 passengers); the total number of pedestrians (357) and bicyclists (74) killed in cities is also shown. Since some pedestrians and bicyclists in cities are killed by vehicles other than cars, this information is not quite sufficient, but it does suggest that the number of pedestrians and bicyclists killed by car accidents in cities may be roughly comparable to the number of killed drivers and passengers and is certainly not much larger. The number of drivers and passengers killed in Paris has averaged 1.7 per year between 2007 and 2009 (F. Prochasson, Préfecture de Paris, personal communication), and in view of the average data for French cities we take the total fatality rate to be about twice as large. EQT (2004) indicates that the number of car-km/day in Paris is about 2.5 million. The 160,000 bicycle trips per day in Paris imply 0.8 million bicycle-km/day if one assumes 5 km per trip. The numbers for Paris in this section imply that the fatality rate per bicycle-km is about (5.3/0.8)/(2∗1.7/2.5)=4.9 times higher than the fatality rate per car-km. In other words, in Paris the avoided car fatalities due to our scenario are small compared to the added deaths of bicyclists.

In view of this situation we consider Amsterdam and Paris as lower and upper bounds, i.e. zero as lower bound for the cost of fatal accidents of our car-to-bicycle mode shift and 105 €/yr per bicyclist as upper bound, and their mean 53 €/yr as central estimate.

4. Results

The steps of the calculations and the results for an individual who switches from car to bicycle are shown in Table 5. The results are plotted in Fig. 2. The calculations for drivers who switch to walking are similar.

For our walking scenario the benefit of PA is 1192 €/yr. The public benefit is only 16.5 €/yr because the trip is half as long as for bicycling. The change in pollution exposure and intake implies a cost of 15 €/yr for the individual. We have not evaluated a possible change in accident risk for walking.

The error bars in Fig. 2 indicate confidence intervals. For the gain from PA these were calculated by repeating the life table calculation with the 95% lower and upper bounds (0.094 and 0.447) of (1−RR) of the DRF for bicycling. For pollution we estimate the confidence intervals according to Spadaro and Rabl (2008). For fatal accidents the error bars indicate the range between the values for Amsterdam and Paris. We do not include the uncertainty of the monetary valuation in these error bars because it affects the costs in the same manner (although for accidents there is an additional uncertainty due to the ratio VPF/VOLY). The reader can readily scale the graph for a different valuation of mortality. For the uncertainty of the latter we estimate that the valuation could be a factor of two higher or lower.

Table 5
Calculations and results for mortality impacts of switch from car to bicycle.

Item	Value	Unit	Explanation
Health gain from PA			Health gain of individual due to physical activity
RR	0.709		Solid circle in Fig. 1
LE gain	1.20	yr	Life table calculation for EU25
Lifetime benefit	52418	€	LE gain × VOLY
Benefit per year	**1310**	€/**yr**	Lifetime benefit/40 yr
Public health gain			Due to reduced emission of pollution
$PM_{2.5}$ emission/km	0.031	g/km	Table 3, average diesel and gasoline EURO4
Length of trip	5	km	One way
Number of trips/yr	460	/yr	2 × 5 trips/week, 52 − 6 weeks/yr
$PM_{2.5}$ emission/yr	71.8	g $PM_{2.5}$/yr	Avoided emissions due to shift to bicycling
Avoided damage cost	458.3	€/kg of $PM_{2.5}$	Table 4, average large cities
Benefit per year	33	€/yr	
Change of individual dose [a]			Due to change in exposure and intake
Concentration	23	μ/m^3	Concentration of $PM_{2.5}$ in street
DRF	0.00065	YOLL/(pers.yr $\mu g/m^3$)	Slope of DRF for mortality due to $PM_{2.5}$
Duration–car	0.25	h/trip	Duration of car trip
Modifying factor–car	1.5		For exposure and inhalation of driver, relative to DRF of general population
Cost–car	4.30	€/yr	Avoided cost, relative to general population
Duration–bicycle	0.33	h/trip	Duration of bicycle trip
Modifying factor–bicycle	3		For exposure and inhalation of bicyclist, relative to DRF of general population
Cost–bicycle	22.9	€/yr	Cost increase relative to general population
Benefit per year	− **19**	€/**yr**	Negative, i.e. cost, of exposure change car–bicycle
Fatal accidents[b]			Increased mortality due to accidents
Accident rate	6.6E − 05	Accidents/yr per bicyclist	Paris
Accident rate	2.5E − 05	Accidents/yr per bicyclist	Amsterdam
Cost/accident	1.6	M€$_{2010}$	VPF
Benefit per year	− **53**	€/**yr**	Average of 0 in Amsterdam and − 105 in Paris Negative, i.e. cost, of risk change car–bicycle

[a] Highly dependent on details of trajectory, could even have opposite sign.
[b] Highly dependent on details of trajectory and behavior of drivers and bicyclists in the city.

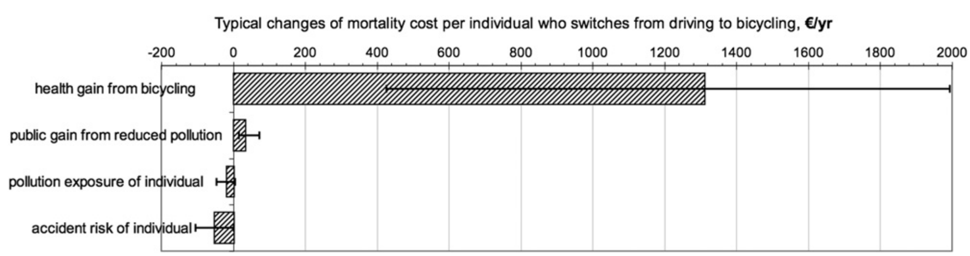

Fig. 2. Results for mortality costs and benefits per individual who switches from car to bicycle for commuting to work (2∗5 km roundtrip, 5∗46 weeks/yr) in large cities of EU. Error bars indicate confidence intervals.

5. Discussion

Despite the uncertainties, and whatever one assumes about the scenarios and the impacts of car emissions, the key conclusions about the health impacts are not affected: by far the most important item is the health benefit due to physical activity. The benefit for the general population due to reduced air pollution is much smaller, and in large cities it is larger than the cost due to changed exposure for a driver who switches from car to bicycle; in small cities or rural zones the public benefit is small or negligible. The exposure change for the individuals who switch implies a loss with our assumptions, but could be a gain if the bicycle can travel on a path with lower pollution. The concern about pollution exposure of bicyclists, often evoked in the context of bicycling in cities, is unfounded when compared to the benefits of the cycling activity; of course, such exposure should be minimized as far as is practical. Accidents can be a more serious problem and more should be done to reduce the risks.

Our results for the effects of pollution are entirely consistent with the site specific calculations of de Hartog et al. (2010) and Woodcock et al. (2009), but they are more general because we have considered many sites. Our estimate of the LE gain due to bicycling is about twice as large as that of de Hartog et al because our life table calculation considers the full steady state benefit, attained by someone who has been bicycling from age 20 to 60. In the near term the benefit is smaller because the risk reduction is applied only for a limited number of years.

So far we have considered only mortality. Had we included morbidity endpoints, the numbers for public and individual air pollution impacts would be about 50% larger according to the DRFs and monetary values of ExternE (2005). Since the health benefits of physical activity span a wider variety of important endpoints, as explained in Section 2.2, the value of the benefit may be increased by more than 50%, but we have no specifics to support this possibility. The cost of bicycle accidents would be very much larger than our numbers, as demonstrated by a detailed investigation of nonfatal bicycle accidents in Belgium by Aertsens et al. (2010). These authors find that the average cost of such accidents is 0.125 € per km bicycled. Applied to our scenario this implies cost of 286 €/yr for the individuals who switch to bicycling.

In addition to health, such a switch can bring several other important benefits, especially reduced congestion and reduced street noise. We have not studied these topics in detail but cite numbers from a recent assessment of external costs of transport in the EU (CE Delft, 2008). In Table 6 we summarize key results of that report for the average damage cost per km. For the sake of illustration in the example below we choose a congestion cost of 0.75 €/km and a noise cost of 0.76 €/km.

In Fig. 3 we show what these numbers imply for our bicycling scenario. Typical average benefits from reduced congestion and noise may well be even larger than the health gain from physical activity. In this figure we have also added the benefit of reduced green house gas emissions, assuming 25 € per tonne of CO_2, reasonable in view of current assessments albeit extremely uncertain and controversial. But compared to the other costs and benefits it is negligible, unless the cost per tonne of CO_2 is very much larger.

Table 6
Average damage cost per km due to congestion and noise of passenger cars in the EU. From CE Delft (2008), Table 7, p. 34 for congestion and Table 22, p. 69 for noise. We use the bold face values for Table 7 and Fig. 3.

Congestion Area and road type	Min.	Central	Max
Large urban areas (> 2,000,000)			
Urban motorways	0.30	0.50	0.90
Urban collectors	0.20	0.50	1.20
Local streets center	1.50	2.00	3.00
Local streets cordon	0.50	**0.75**	1.00
Small and medium urban areas (< 2,000,000)			
Urban motorways	0.10	0.25	0.40
Urban collectors	0.05	0.30	0.50
Local streets cordon	0.10	0.30	0.50
Noise [a] Time of day	Urban	Suburban	Rural
Day	**0.76**	0.12	0.01
Range	(0.76–1.85)	(0.04–0.12)	(0.01–0.014)

[a] For noise the lower limit of the range is based on dense traffic situations, the upper limit on thin traffic situations. Central values are for the predominant traffic situation in the respective regional cluster: urban: dense; suburban/rural: thin.

To illustrate how our results can be used for evaluating transport policies, let us take the example of the Vélib Program in Paris. Vélib is a system of rental bicycles, comparable to similar systems that have been implemented in recent years in other cities of the EU. At the present time there are about 20,000 Vélib bicycles in Paris, and the total cost of the program is currently about 64 M€/yr. Per bicycle that amounts to 3200 €/yr, very expensive because of high repair and maintenance costs.

To see whether such high cost can be justified, one would need to know how many Vélib users have switched from which transport mode. In addition one should consider how many other bicyclists have made the switch to bicycling because of seeing the example of Vélib riders. That sort of information can only be obtained by surveys of individual bicyclists. Unfortunately we do not have such data. Furthermore, many bicyclists in Paris switched from public transportation to avoid congestion during rush hour, and so we would also need an estimate of the impacts of commuting by underground and/or bus. In Paris there is another factor that complicates an assessment of the benefits of the Vélib program by itself: the city has been creating bike paths and designated lanes for buses by reducing the space available for cars, thus putting pressure on people to switch from car to public transportation or active transport.

Obviously we cannot do a meaningful cost-benefit analysis. But at least we can try to obtain an upper bound on the benefits by noting that the total number of one-way bicycle trips (Vélib and private) in Paris is about 160,000 per day, and very roughly half of them use Vélib. As a gross simplification, let us assume that each Vélib bicycle is used for the equivalent of two round trips per day of our scenario, in other words, that there is the equivalent of 40,000 commuters who make the switch from car to Vélib; in reality the number of Vélib users who are former drivers is probably smaller. Multiplying the costs in Fig. 3 by 40,000 we obtain the results in Table 7. Thus the total benefit is probably smaller than 176.9 million €/yr, i.e. less than 2.8 times the cost. The benefit is greater than the cost if Vélib has induced a net shift of at least 14,500 drivers to bicycling.

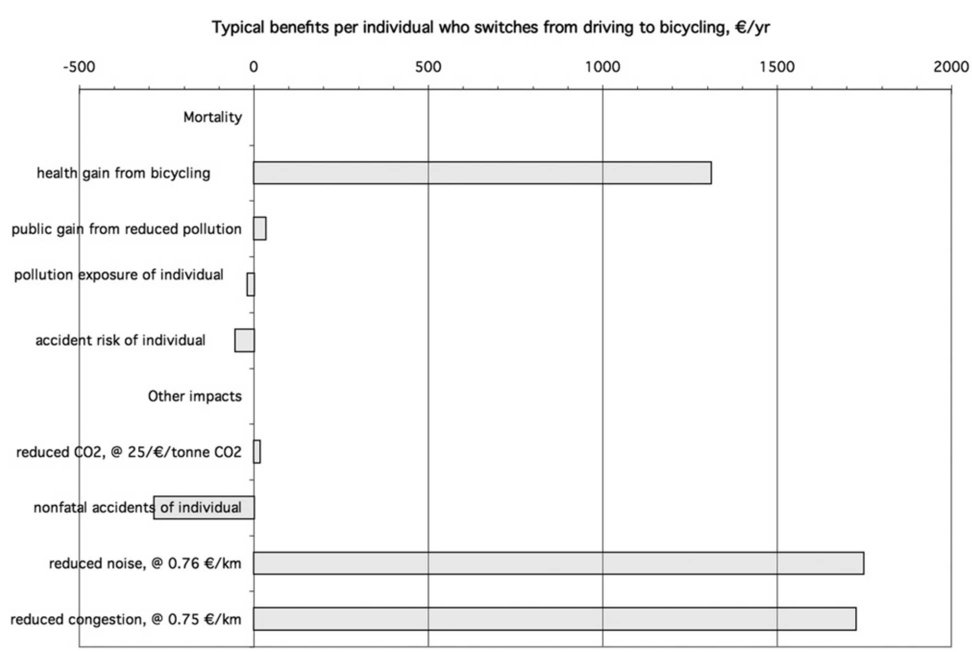

Fig. 3. Comparison of mortality costs and benefits with other impacts, for our bicycling scenario.

Table 7
Upper bound of benefits of Vélib bike sharing program in Paris.

Item	Amount, M€/yr
Health gain from bicycling	52.4
Public gain from reduced pollution	1.3
Pollution exposure of individual	−0.7
Fatal accidents	−4.2
Nonfatal accidents	−11.5
Reduced CO_2 emissions	0.6
Congestion	69.0
Noise	69.9
Total benefit	**176.9**

6. Conclusion

We have carried out a detailed analysis of the mortality impacts of a shift to active transport, using specific scenarios that are reasonable but can readily be modified by the reader. Despite large uncertainties one can firmly conclude that by far the most important item is the health benefit due to the physical activity. The benefit for the general population due to reduced air pollution is much smaller, but in large cities it is larger than the cost due to changed exposure for a driver who switches from car to bicycle. For a mode shift in rural areas the public benefit is very small. The exposure change for the individuals implies a loss with our assumptions, but could be a gain if the bicycle can travel on a path with lower pollution. In any case the benefits of bicycling completely overwhelm any concern over pollution exposure of bicyclists. Of course, such exposure should be minimized, for example by not riding a bicycle behind a bus or truck and by placing cycle lanes in less trafficked streets. Accidents are a more serious problem and more should be done to reduce the risks.

The conclusions about the relative magnitude of the effects also hold for individuals who switch from driving to walking. Incidentally the role of physical activity (walking to the station, standing, climbing stairs to the subway) is not negligible when people switch from driving to public transportation and the associated benefits may well outweigh the increased exposure to PM that has been observed in subways and many buses.

In addition to this detailed discussion of mortality impacts, we have also cited numbers from the literature to indicate the magnitude of other benefits of a shift to active transport, especially reduced noise and congestion. Our results can be applied to evaluate proposed policies or projects, for example public programs for the rental of bicycles (now implemented in many European cities) or projects to create more bicycle paths, if one can estimate the number of individuals who shift their transport mode.

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