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Review

Wellbeing Impacts of City Policies for Reducing Greenhouse Gas Emissions

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Abstract: To mitigate climate change, city authorities are developing policies in areas such as transportation, housing and energy use, to reduce greenhouse gas emissions. In addition to their effects on greenhouse gas emissions, these policies are likely to have consequences for the wellbeing of their populations for example through changes in opportunities to take physical exercise. In order to explore the potential consequences for wellbeing, we first explore what 'wellbeing' is and how it can be operationalized for urban planners. In this paper, we illustrate how wellbeing can be divided into objective and subjective aspects which can be measured quantitatively; our review of measures informs the development of a theoretical model linking wellbeing to policies which cities use to reduce greenhouse gas emissions. Finally, we discuss the extent to which the links proposed in the conceptual model are supported by the literature and how cities can assess wellbeing implications of policies.

Keywords: climate change; greenhouse gas emissions; cities; wellbeing

1. Introduction

This paper is part of URGENCHE (Urban Reduction of Greenhouse Gas Emissions in China and Europe), a European Commission funded project to assess the health and wellbeing implications of city policies for reducing greenhouse gas (GHG) emissions. The assessment is based on scenarios within seven case study cities and aims to identify the effects that municipal housing, transport and energy measures to reduce GHG emissions would have on health and wellbeing by the year 2020. While health related results are beginning to be appear elsewhere [1], this paper focuses on wellbeing firstly because, despite being less well understood than health, wellbeing is a desirable goal in itself [2] and secondly it is critical for future wellbeing that we preserve an environment meets basic needs such as water and clean air [3]. Climate change threatens this and even where basic needs can be met it is likely to increase psychological stressors through, for example, unpredictable weather patterns and migration [4]. To implement mitigation policies the agreement of the people is needed. Consensus is more likely if current wellbeing is not compromised and if there is a shared understanding of the possible co-benefits compared with dis-benefits of policies [4,5].

WHO Regional Office for Europe's proposed definition is that wellbeing "comprises an individual's experience of their life as well as a comparison of life circumstances with social norms and values" [6]. In addition to academics, "wellbeing" is of interest to charities [7], non-governmental organizations [8,9] and governments [10] in order to understand how society is "doing" [11]. Given the aim of URGENCHE is to provide quantitative estimates of consequences of implementing GHG reduction policies, wellbeing is primarily operationalized here through scales and indices rather than through qualitative work.

The project's empirical limitation to the real actions implemented in cities has meant that URGENCHE assessments of wellbeing are restricted to the GHG interventions under consideration by the project cities: it therefore does not include urban planning polices such as creating green space, or increasing housing density as they were not chosen by the URGENCHE cities as part of the project. However, this does not indicate that such policies would not have wellbeing effects.

The objectives of this paper were firstly to develop a conceptual framework of wellbeing relevant to greenhouse gas reduction policies and secondly to operationalize the introduced concepts in order to guide the study of the effects of GHG policies on wellbeing. The conceptual model emerged from both *a priori* and *a posteriori* processes: from the authors' previous experience of the literature in this area and their experiences of working with cities for URGENCHE and also from evidence collected through our compilation of literature (details provided later) undertaken specifically for the project; thus the authors concur with Williamson that the distinction between the two processes is superficial [12].

2. A Conceptualization of Wellbeing Relevant to Greenhouse Gas Reduction Policies

In this section we make a very limited introduction to the concept of wellbeing in order to introduce the reader to the concept. If the reader is interested in a more critical study of the concept we suggest

that they peruse some of the following references [13–18].

The concept of wellbeing is currently under discussion and development but it is generally recognized that it involves subjective and objective components [6,19–22]. The subjective aspects of wellbeing involve the first part of the WHO Regional Office for Europe definition-an individual's experiences [6] including "psychological functioning and affective states" [19]. Objective wellbeing involves
"a comparison of life circumstances with social norms and values" [6]. Thus wellbeing can be seen as enacted on both an individual subjective level and a social objective level. In addition to the subjective/objective dimension, wellbeing is also theorized in terms of hedonic/eudemonic dimensions: "hedonic" wellbeing involves happiness, pleasure and enjoyment where wellbeing is achieved by avoiding pain and seeking pleasure, and "eudemonic" wellbeing which is achieved through finding purpose, meaning and fulfillment [20,21,23].

WHO's 1948 Constitution defines health as "a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity" [24] and thus views disease and infirmity as one end of a spectrum and wellbeing as the opposite end; thus health is both absence of disease and presence of wellbeing. It appears that health is both a determinant and an outcome of wellbeing or they are mutually constitutive factors [19]. The definition also reflects the Cartesian philosophical idea that the mind and the body represent different functioning systems [25] and thus physiology (or physical health) can be separated from subjective wellbeing. Despite the shortcomings of this approach [25], for clarity we will use "health" in this restrictive sense, referring to absence of disease and directly measurable outcomes such as life expectancy. Although the WHO definition of health (which encompasses wellbeing) is from 1948, it is only in the last three decades that discussion on the need to measure different aspects of non-physical health and wellbeing has gained prominence.

The term "wellbeing" is used in association with "positive mental health" [26,27]. In our conceptual model we have therefore considered studies that measure mental health to be measuring wellbeing; however it must be noted that wellbeing is more than just absence of psychological distress [22,28,29]. The WHO Regional Office for Europe argues that adverse outcomes resulting from lack of wellbeing are mainly depression and other mental illnesses and thus subjective wellbeing should be measured because negative outcomes lead to costs for health services [19]. Mental disorders involve the inability to manage thoughts, emotions, behaviors and interactions with others and can be caused by social, cultural, economic, political and environmental factors; these include national policies, social protection, living standards, working conditions, and community social supports [30]. Mental health, and in particular depression, has been measured in a variety of contexts, both on a continuous scale, other times as a dichotomy (such as depressed *vs.* not depressed). Self-assessed health tends to reflect respondents' mental health and thus is also an indicator of mental health [31]. Thus, mental health can be measured in a variety of different ways. Some studies have however used "wellbeing" itself as an outcome rather than mental health but again there are many available measures.

There is no single ideal wellbeing measurement [19] and through a critical examination of the operationalization of wellbeing by various measures, we intend to illustrate further the concept of wellbeing through issues that arise. The review of wellbeing measures below shows that many current measures reflect different understandings of wellbeing and often confuse objective and subjective

wellbeing; ideally self-complete scales assess subjective wellbeing, while wellbeing indices include objectively measured environmental and personal conditions which are likely to lead to high subjective wellbeing (There is considerable discussion on how indices and scales should be differentiated. For the purposes of this article, a scale is multiple items usually measuring one factor, using a common set of responses (e.g., agree strongly, agree slightly, disagree slightly, disagree strongly), whereas an index is where multiple indicators are amassed). Although some have argued that a fuzzy definition of wellbeing is helpful in that it allows people to use wellbeing for their own purposes [32], we suggest that conflating objective and subjective wellbeing lays the field open to criticism as the direction of causality is difficult to infer.

In this paper, we seek to help academics and policy makers choose how to measure subjective and objective wellbeing as potential outcomes of their policies and thus we review subjective wellbeing measures and refer to some of the available general objective wellbeing measures in advance of concentrating on measure of objective wellbeing that are more likely to be affected by GHG reduction policies. Our review also informed the development of our conceptual model.

2.1. Subjective Wellbeing Measures

There are many subjective wellbeing scales (see [33–35] for more detailed overviews) and we have space here only to discuss five examples which are intended to illustrate the breadth of possible measures and help the reader to reflect upon what they believe the concept of wellbeing to be. We encourage the reader to think about more than just statistical validity and to instead consider carefully the items used in the scale. Firstly we discuss the WHO-5 scale which, after careful consideration of the items found in a number of scales, was used in the development of our conceptual model of environmental impacts on wellbeing. Other approaches to wellbeing, such as satisfaction, of course have merit and have been used in peer reviewed and well-received studies of wellbeing so we continue by describing selected alternative measurement tools and their approach to define wellbeing, illustrating the diversity of measures and concepts developed and their strengths and weaknesses.

The WHO-5 Wellbeing scale [36] was developed specifically to measure wellbeing. It has been translated into many languages and has been successfully statistically validated in a variety of populations [37–41]. It is practical to use, consisting of only five questions. Respondents are asked to rate their wellbeing on a six point scale over the last two weeks, thus it is not just measuring momentary feelings. The five items capture hedonic aspects of wellbeing (cheerfulness and good spirits, the abilities to relax, feel rested and be active) and the eudemonic aspect in "experiencing life as full of interest".

A possible disadvantage of WHO-5 is that it was originally developed to measure wellbeing in diabetes patients and thus the wellbeing measured could be "wellbeing despite disease" which could be problematic in a healthy population. The recognition that people with physical health issues can experience high levels of wellbeing is of significance in itself and lends credence to the notion that wellbeing is more than just one end of a spectrum with disease at the other end. Measures of physical health are often included in objective wellbeing indices, as is socioeconomic status; those measuring objective wellbeing without also measuring subjective wellbeing are making assumptions that those in

poor physical health and socioeconomically disadvantaged groups are not experiencing high levels of wellbeing, but both these groups can report good levels of subjective wellbeing [40,42].

Nevertheless the WHO-5 scale has however now been successfully used in a variety of other settings in addition to diabetes research [43–45]. Notably, in the light of GHG reduction policies, the WHO-5 is included in the European Quality of Life Survey 2012 [45], which enables the measurement of linkages between external conditions (such as social, occupational, and environmental domains) and subjective wellbeing [27].

Another well-known successfully statistically validated scale measuring subjective wellbeing is the WEMWBS Warwick-Edinburgh Mental Well-Being Scale [26,46,47]. The fourteen WEMWBS items allow the scale to widen the definition of wellbeing but are more onerous for respondents to complete. There is now a shorter seven item version available (SWEMWBS or short WEMWBS) [48]. The items are "I've been feeling optimistic about the future", "I've been feeling useful", "I've been feeling relaxed", "I've been dealing with problems well", "I've been thinking clearly", "I've been feeling close to other people" and "I've been able to make up my own mind about things". Thus the SWEMWBS is more focused on cognitive, rather than affective, aspects than WHO-5 and deliberately includes relationships with other people as part of wellbeing itself. There is consequently the possibility that this formulation of wellbeing may be less likely to apply universally to all people because perceptions of social ties with other people is, at least to some extent, culturally patterned. Both the WHO-5 and SWEMWBS phrase all items positively, in contrast to scales measuring mental illness, but the SWEMWBS scale refers to concepts that are not positive such as "dealing with problems" and "having to make decisions" which may not carry positive valence.

A different approach focuses on satisfaction with life, which can be operationalized as a single item or as a tool covering various elements. Observation of the extensive use of life satisfaction in international surveys [34] and concerns about the theoretical and empirical underpinnings of the links between hedonism and eudemonism and health [49] (which have some potential to be addressed in this paper or by theoretical and empirical research inspired by this paper), led the WHO Regional Office for Europe to adopt a single item on satisfaction with life as the core indicator for monitoring subjective wellbeing in the newly established health policy (Health 2020) [50]. An example of a more complex tool using multiple items is the successfully statistically validated "satisfaction with life scale" developed in the US [51,52]. Diener's definition of wellbeing defines subjective wellbeing as "how people evaluate their lives" [53,54]. Thus the five items ("In most ways my life is close to my ideal", "the conditions of my life are excellent", "I am satisfied with my life", "So far I have gotten the important things I want in life" and "If I could live my life over, I would change almost nothing") are all about evaluation rather than general feelings. The comparative element in this scale may be too close to objective wellbeing as it is likely to shift a person's thoughts towards their life conditions rather than feelings: it would be hard for a person with low socioeconomic status or poor social relationships to acquire a high score (see [42]). Furthermore as a term, "satisfaction" is highly subjective and charged with different meanings according to context; for example the rating "satisfactory" has recently changed from being acceptable to "requires improvement" for English school inspections [55].

The developers of a Dutch subjective wellbeing scale, the SPF-IL scale [56] noted that people tend to assess wellbeing affectively (as in the WHO-5) and cognitively (as in SWEMWBS) and attempts to focus respondents answers to reflect both by asking respondents about their experiences rather than

general statements. These authors used the theory of Social Production Functions to develop items intended to measure how well respondents reach goals of affection, behavioral confirmation, status, comfort and stimulation. However, the scale encompasses fifteen items, and in the validation study there were many missing values for the status items, suggesting this scale did not appeal to respondents.

Finally, the UK Office of National Statistics (ONS), uses four experimental questions on subjective wellbeing: general life satisfaction; feeling that actions are worthwhile; happiness yesterday; and anxiety yesterday [57]. These questions cover both hedonic and eudemonic aspects of wellbeing. The four questions are analyzed separately rather than being combined into a scale. Eleven point scales are used for responses. However, generally the upper limit for Likert scale responses is seven [58] due to limitations in human's ability to visualize larger numbers [59]. Another possible criticism is that the questions only consider as far back as yesterday whereas the WHO-5 scale answers over a longer time period (two weeks). Subjective wellbeing when measured over time is surprisingly stable [60]. To be certain to be measuring this more stable concept the WHO-5 scale acknowledges the temporal context of wellbeing by asking about experiences over the duration of a two week period. Thus the ONS questions may reflect particular life events rather than a more stable feeling.

These five examples of subjective wellbeing scales provide some understanding and evaluation of the breadth of approaches to the quantitative measurement of subjective wellbeing. Whether wellbeing is primarily emotional or cognitive, whether relationships with other people are a predictor of or part of wellbeing and the universality of the wellbeing measured is likely to vary with the scale used. Only the WHO-5 and ONS clearly include both hedonic and eudemonic aspects. The cognitive assessments of SWEMWBS and evaluations of the "satisfaction with life" scale may occur from a person either reflecting that they have positive life circumstances which lead to (or result from) subjective wellbeing rather than subjective wellbeing itself (as measured by the WHO-5 scale). However, irrespective of the conceptual differences of the tools, there is little likelihood that cities regularly collect data on any of these; nevertheless the WHO-5 has the additional advantage of having been included within the EQLS 2012 [45], enabling associations between wellbeing and a variety of other dimensions to be tested, thus making it a pragmatic choice for theory development which can then be empirically tested.

2.2. Objective Wellbeing Indices

There has been growing interest in developing "objective wellbeing" [20] or "livability" [61] measures often at a national scale [6,30]. A range of projects, such as WHO Regional Office for Europe's consultation on targets and indicators for wellbeing [49,62] and indices attempt to measure objective wellbeing, such as the OECD Better Life Index [63] and the Oxfam's Humankind Index [7]. National measures are also being developed, for example, by Gallup (US) [64], Istat (Italy) [65], Health Utilities Inc. (Canada) [66], INSEE (France) [67] and ONS (UK) [68]. The Dutch "Leefbaarometer", a survey run every five years, offers detailed information on a list of wellbeing issues at a zip code scale [69,70]. Commonly the indices include health, health related behaviors, sustainability and environment, socioeconomic status and social support. Social factors are more to the fore in comparison to the individualistic style of items included in subjective wellbeing scales. More detail is now provided on sustainability and environment, socioeconomic status and social support with the exception of health and health behaviors because health has been discussed previously and physical

health, although relevant, is outside the scope of this paper.

2.2.1. Sustainability and Environment

Often progress is measured in terms of GDP [8], either because wealth is seen as an end as itself or because economic wellbeing is recognized as important [32]. Wellbeing is not just about economics and striving to increase wellbeing in general, rather than GDP, has been argued to lead to a more sustainable future [8]. However, there is potential tension between wellbeing and sustainability [71]: either there is a compromise on wellbeing now to improve wellbeing for future generations or there is no compromise now but there but there will be severe reduction in wellbeing for future generations [72]. Wellbeing thus needs to be considered temporally (regarding future catastrophe) and spatially (regarding environments we inhabit).

Temporal conceptualizations of wellbeing take into account changes in wellbeing over time and the difference between short and long term goals [73]. It is an issue for city planners that citizens may think of their own short term wellbeing rather than the wellbeing of future generations when they evaluated GHG reduction options [74]. It may also be hard for people to change their life patterns which provide short term comfort, such as car use, for long term environmental sustainability [75].

Because many of the factors affecting wellbeing are spatially structured (as for example they involve contextual variables pertaining to local communities, such as cohesion), the environment and location-specific factors have a much larger influence in determining wellbeing than previously thought for example through natural environment characteristics, services available, and congeniality and socioeconomic status of the population [76]. It has been argued that these are as important as individual socio-economic or demographic factors [77]. The significance that respondents attribute to wellbeing and wellbeing scores can vary depending on the cultural and political context [6,78,79] but nevertheless there are sufficient synergies for comparisons between areas to be valid [42]. A spatial conceptualization can be described by maps or a lived experience of a "place" [80]) and need to bear in mind geographic scale that ranges from the individual to more aggregate levels [81,82]. Objective social indicators collected for well-defined administrative units or areas are unlikely to represent the territorial base of an individual's wellbeing [83]. Neighborhood satisfaction, for example, will depend on the effective space "inhabited" by an individual, and be meaningful in relation to that space, rather than administrative units [84].

Thus, consideration of climate and environmental conditions is critical when analyzing objective wellbeing [77,85] and therefore, many objective wellbeing indices include parameters related to the provision of the population with adequate environmental services and conditions (e.g., the Dutch Leefbarometer covers housing, noise and green spaces, the OECD Better Life Index includes an environmental component covering air pollution and water quality and a housing component covering rooms per person and dwelling facilities, and the Humankind Index from Oxfam covers green spaces, clean and healthy environments, and having an affordable and decent home).

The literature of sustainability and wellbeing has been drawn upon in the conceptual model in the second part of this paper—although, for simplicity, the model underrepresents concepts of time and space.

2.2.2. Socioeconomic Status

One common component of objective wellbeing that has relevance for cities but warrants particular discussion is socioeconomic status (SES). There are concerns that a focus on the fuzzy concept of wellbeing may reduce the priority to decrease inequalities [86]. Nevertheless policies to reduce disadvantage should also improve wellbeing because low SES is associated with lower subjective wellbeing whereas affluence, however, is not strongly associated with higher subjective wellbeing especially when subjective wellbeing is measured in terms of stable affect rather than in terms of satisfaction [42]. This may be because materialism is associated with poorer wellbeing [72]. Thus in the urban context, it is important to be able to differentiate the wellbeing effects of policies on citizens of different socioeconomic levels.

Given that residential choices are often affected by socioeconomic status, there are spatial injustices in access to services, geographical variations in standards of living and exposure to pollution or noise, and discrepancies in access to therapeutic landscapes and health-promoting urban features [87]. Furthermore there needs to be a balance between direct effects of a policy on wellbeing and indirect effects for example through economic growth [88], as a factory may reduce health and wellbeing of the local community through air pollution but may increase health and wellbeing through employment.

2.2.3. Social Relationships

Social relationships are often viewed by social scientists through the concept of social capital. Social capital involves, in addition to positive informal social relationships, participation in clubs and voluntary associations, voting patterns and social trust [89]. Social capital can be conceived of as a kind of aggregate level of eudemonic wellbeing (and perhaps overlapping with the Chinese concept of "harmonious society" [90]).

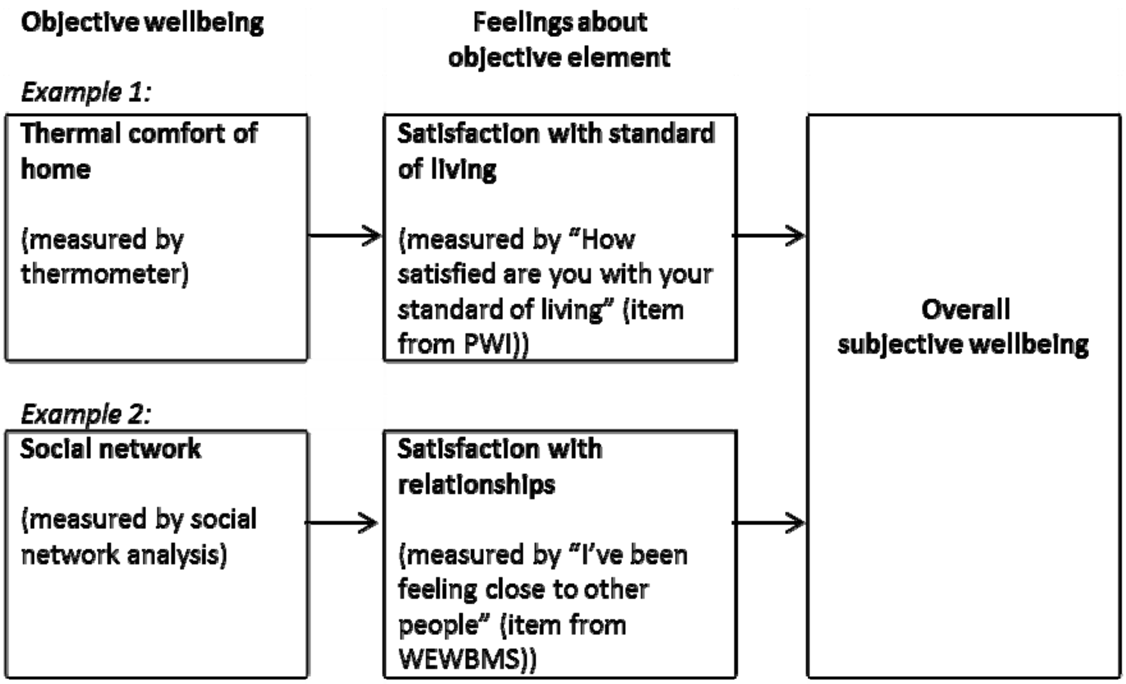
2.3. Combined Measures

Some instruments, for example WHO QOL-BREF [91], EUROHIS-QOL [92] and the Happy Planet Index [8] combine subjective and objective wellbeing measures and subjective wellbeing is often included as one measure within what are ostensibly objective wellbeing indices (e.g., [65]). This is a problematic procedure because the two are very different concepts and should be kept separate in order to study the intricacies of the relationships between them. Similarly the Personal Wellbeing Index [93,94], although often described as a subjective wellbeing index, has a domain based approach to wellbeing (it asks about levels of satisfaction with a list of specific items (standard of living, health, achievement, relationships, safety, community and future security)). Statistical validation of the PWI shows only moderate correlation between domains [94]. Thus a domain approach may be unsatisfactory as an attempt to measure global wellbeing. A global measure is important because generating a comprehensive list of contributing domains is difficult and also domains on the list, are likely to change over time and even if they remain relevant their importance may change [21].

Furthermore given the PWI is asking about satisfaction with various domains, it is arguably measuring a concept which is on the pathway moving from objective wellbeing to subjective wellbeing rather than subjective wellbeing itself (Figure 1). The three steps on the pathway (illustrated in

Figure 1) are cognizant with the three policy relevant accounts of wellbeing: objective lists, preferences satisfaction and mental states [21,95]. Two domains are provided in Figure 1 as examples of differentiation between 'pure' objective wellbeing, "pure" subjective wellbeing and measures between these two poles. The example domains are thermal comfort of the home and social networks together with examples of measurements. Objectively measurable externalities are evaluated internally and then an overall feeling of subjective wellbeing is likely to arise from the merging of various domains. This pathway is relevant to our topic because for a GHG reduction policy to have a positive effect on wellbeing it will need to have a positive effect on people's feelings as well as on objective measures. For example a policy that reduces car use may increase active mobility and reduce pollution, but it is not necessarily the case that those affected will feel positive about the forced change.

Figure 1. The continuum between objective and subjective wellbeing: an example with thermal comfort and social networks.



In order to operationalize the concept of wellbeing for the purpose of URGENCHE research on urban policies for GHG emission reduction, we summarize our review of subjective and objective wellbeing measures: subjective wellbeing can be measured in a short scale which includes hedonic and eudemonic items but not evaluative and cognitive items whereas objective wellbeing should be measured in terms of tangible independently observed characteristics such as medical conditions, socioeconomic status and characteristics of the environment. From our review of subjective wellbeing indices we conclude that the WHO-5 is a good basis for understanding subjective wellbeing because firstly it does not include feelings about objective elements, secondly it is a global measure rather than domain based, thirdly it is feelings based rather than cognitively based and fourthly it has already been used in a European wide survey [45] of environmental and social dimensions and wellbeing as demanded by the URGENCHE project.

Our review of objective wellbeing indices suggests that socioeconomic status, sustainability, relationships and physical health are important aspects but objective wellbeing will need to be defined

further for our purposes in the conceptual model. In order to develop this analysis of quantitative measures of wellbeing in the context of urban GHG policy, it is necessary to develop a conceptual model which integrates these approaches.

3. Development of the Conceptual Model within Policies for Reducing Greenhouse Gas Emissions

A conceptual model connecting subjective wellbeing, through objective wellbeing, to the potential results of GHG reduction policies is provided in Figure 2. Our conceptual model was theory driven using the wellbeing framework of subjective *vs.* objective wellbeing and hedonic *vs.* eudemonic wellbeing and additionally informed by the overview of the literature on concepts relevant to greenhouse gas emission reduction policies and wellbeing described later. The arrows indicate suspected relationships, likely effects and potential consequences which are often implicit assumptions made by policy makers and academics but have not been previously articulated. The boxes on the left represent examples of urban policies applied to mitigate climate change; the hexagons in the middle represent effects of the policies on objective wellbeing, and in the oval on the right there are facets of subjective wellbeing as measured by the WHO-5 index. The purpose of the conceptual model is to guide cities as to the areas they need to think about when considering how a particular policy could affect wellbeing, but also suggests an analytical research framework for quantifying the potential wellbeing impacts of such policies. There is evidence, or at least discussion on each pathway in the academic literature, particularly the literature on sustainability and wellbeing [72], but more work is needed on the relative strengths of associations.

3.1. Climate Change Policies in the Conceptual Model

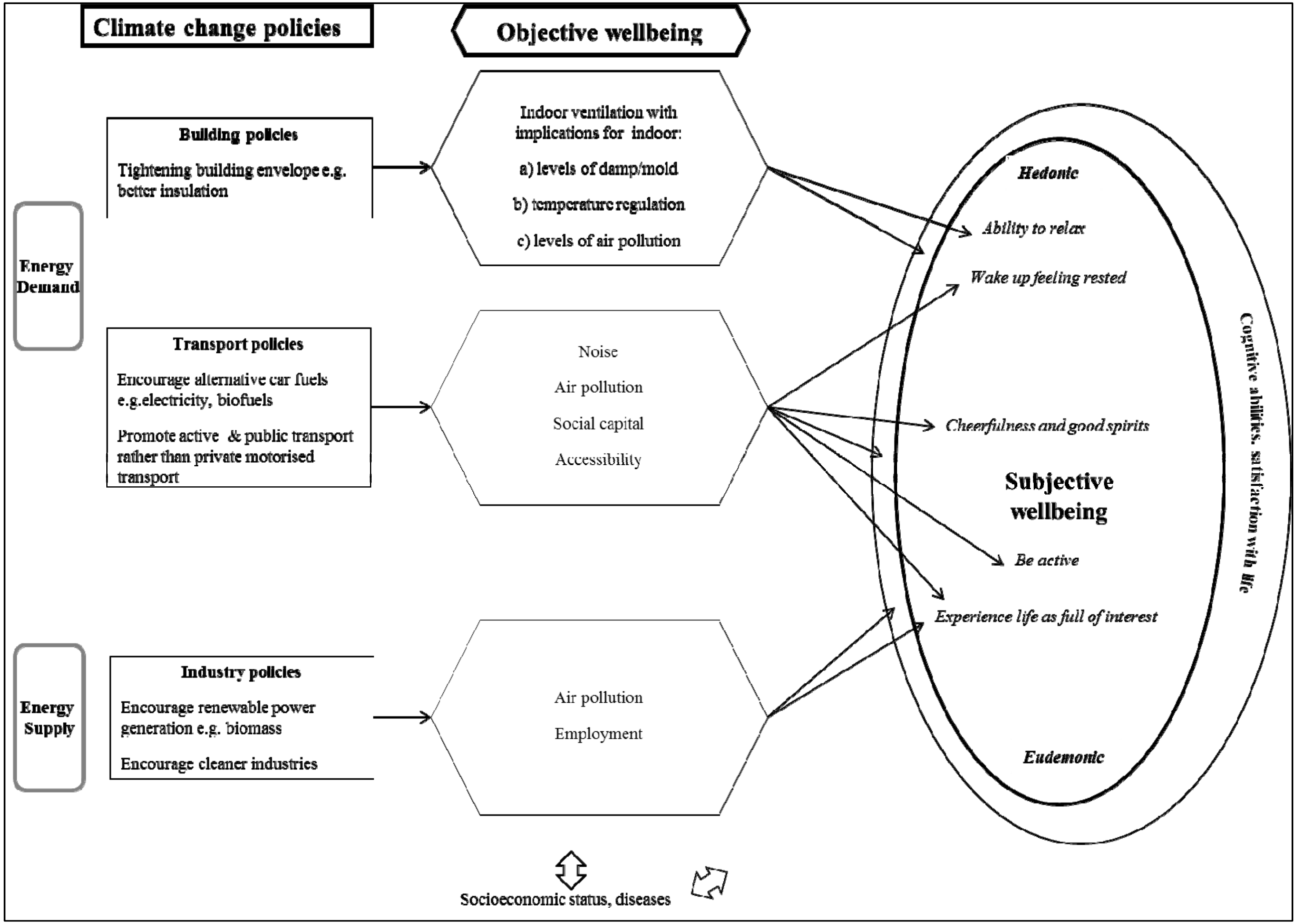
On the left of Figure 2 are some of the GHG reduction building, transport and energy generation policies that cities in URGENCHE wanted to include in modelling. All GHG reduction policies considered by cities are aimed either at energy supply, for example biomass production, or reducing energy demand, for example through tightening the building envelope.

More context is now presented on the potential effects of and linkages between the GHG reduction policies and wellbeing. The following two sections describe the central part of the conceptual model (Figure 2) on how urban policies may affect environmental dimensions relevant for wellbeing. The effects are described in the order that they appear in the model; however many are strongly interconnected.

3.2. Objective Wellbeing Effects of GHG Reduction Policies in the Conceptual Model

In this section, objective wellbeing effects of policies on buildings, transport and industry are discussed.

Figure 2. Conceptual model of some example policies to reduce greenhouse gas emissions and wellbeing.



3.2.1. Building Policies and Objective Wellbeing

Housing policies on tightening building envelopes and improved insulation are likely to have positive implications for thermal comfort. However, studies have shown that a one-sided focus on energy saving without adequate consideration of ventilation rates may increase indoor pollution, dampness and mold growth, negatively affecting indoor environmental conditions [96,97]. Indoor air quality, dampness and mold growth are measurable conditions that, in addition to health, can also affect comfort and feelings of wellbeing [98].

Another potential pathway linking housing and energy efficiency with wellbeing could be the budget savings households would make by reduced heating bills, releasing this money for other household needs and thus affecting wellbeing indirectly.

3.2.2. Transport Policies and Objective Wellbeing

Transport policies are thought to affect wellbeing through ease of access to daily life destinations such as work, education, recreation and consumption, through the benefits of mobility itself (related to social relationships/social capital and physical activity), and externalities such as air and noise pollution [99].

Adequate access to a variety of destinations has been found to be important for objective wellbeing in terms of social capital, work opportunities and physical activity [99]. However, discouraging the use of private transport may reduce the accessibility of some destinations such as employment [100], cultural activities [101], green space and other destinations that engender physical activity and places to socialize [102]. Improvements in public transport and designing walkable neighborhoods [103,104] may mitigate this to some extent. However, it should be noted that the effects of accessibility may be superseded by socioeconomic status as disadvantaged areas in some cities may have many destinations on their doorstep whereas wealthy households who can afford one car per adult may commute in from great distances [105].

Social capital can be developed through encouraging active or public transport, for example through street connectivity, so that people spend more time in their local areas and through interacting with people on public transport and conversely people are more likely to walk if they have higher levels of social capital [106–108].

Finally, high numbers of petrol and diesel powered vehicles in urban settings may cause annoyance from air pollution and noise [72,109–111]. Promoting electric cars and active transport rather than petrol/diesel transport may mitigate the local environmental pollution associated with petrol and diesel use in urban traffic and reduce noise at the low speeds generally found in cities (although quieter cars travelling at speed may result in more accidents) [112].

3.2.3. Industry Policies and Objective Wellbeing

More sustainable and effective energy generation and consumption patterns within the urban industrial sector could significantly reduce not only GHG emissions, but also the emission of air pollutants within a city. Similar to the traffic-related local pollution, this could be expected to have an impact on both health and wellbeing.

Changes in energy supply and improved production technologies might not only affect energy efficiency alone, but also increase general productivity. This may affect productive and industrial activities and result in a net growth or decline in employment [113]. For example, employment decline could occur in cities where local heavy industry and energy production is outsourced to reduce CO_2 emissions within the city itself; while new employment options could be generated through green economy investments in e.g., renewable energy technology or sustainable production [114]. It is possible that low-carbon economies may enable more jobs to be created than lost [115].

In summary, housing, transport and industry policies on reducing energy consumption and GHG emissions may have direct and indirect impacts on objective wellbeing though modification of housing conditions, air quality, social capital, accessibility and unemployment. The examples of employment and potential indoor problems show that the effects are not exclusively beneficial, indicating that such policies may have caveats and negative outcomes also. However, irrespective of the evaluation of the effects as positive or negative, these environmental dimensions of objective wellbeing will have further implications for individual subjective wellbeing as discussed below.

3.3. Subjective Wellbeing Effects of GHG Reduction Policies in the Conceptual Model

The oval representing subjective wellbeing in the model provides an assessment of a "pure" form of subjective wellbeing, as the WHO-5 wellbeing score does not refer to intermediate domains such as health, personal relationships, environment or thought processes or the subjective and situated notion of satisfaction. The five WHO-5 items are included as part of wellbeing and graded by their reflection of hedonism and eudemonism. We suggest that building policies are perhaps more likely linked to hedonism and transport and industry to eudemonism in the following discussion.

3.3.1. Building Policies Implications Subjective Wellbeing (via Housing Conditions)

There is extensive literature on the importance of home and its meaning on people's lives [116]. The ideal home may be a place of comfort to enable inhabitants to rest and relax. If housing conditions are poor—be it due to inadequate thermal comfort, dampness and indoor pollution or other factors-then a dwelling is less comfortable and it may be harder to relax [117]. This also applies to noise which may be generated by transport but largely affects people at home and strongly affects residential satisfaction, which is considered a component of overall life satisfaction and wellbeing [118], thus noise may a reduce the ability to relax and rest and to wake up feeling rested. Various studies have attempted to measure the extent to which noise causes "noise annoyance" or sleep disturbance as potential intermediary factors between noise and wellbeing [118–124].

3.3.2. Transport Policy Implications for Subjective Wellbeing (via Active Transport, Social Capital and Air Pollution)

The objective wellbeing measure of accessibility and the greenhouse gas reduction policy of encouraging active transport are linked to the subjective wellbeing aspect of being active and realization of personal interests [125]. However, they potentially work in opposing directions, as encouraging active transport engenders increased physical activity (which is positive for good spirits

and being active) [101] but reducing the use of other modes of transport could reduce the available venues for such activity to take place.

The same is likely to apply to increased interpersonal contacts and social capital as a potential consequence of more active and public transport, which is likely to interact with cheerfulness and good spirits as well as being active and experiencing life full of interest [126–128]. However, unsafe neighborhoods where people are afraid to be out alone or after dark may counteract such a positive wellbeing effect and actually restrict the ability to relax, to be in good spirits, and to be active within the local neighborhood area [129].

However, no trade-offs would be expected for noise and air pollution, as these should be reduced by the GHG transport interventions and thus improve subjective wellbeing without negative side effects. Perceived levels of noise and air pollution are associated with life satisfaction and happiness [130,131]. More directly air pollution from SO_2, NO_2, $PM_{2.5}$ and PM_{10}, has been found to be associated with mental health [132] in addition to established detrimental physical health effects [110].

3.3.3. Industry Policy Implications for Subjective Wellbeing (via Air Pollution and Employment)

In addition to reducing air pollution from transport, reducing air pollution from industry is likely to have beneficial effects on wellbeing [133] and additionally creation of a cleaner environment may be evidence of responsive governance reducing feelings of powerlessness and stigma among nearby residents [134,135].

The other objective wellbeing effect of industrial change through complying with GHG reduction policies identified is employment opportunities. Becoming unemployed is associated with poor mental health which tends to improve after regaining employment [136]. This is likely to reflect the "eudemonic" aspect of wellbeing [137] (activity level, experiencing life with full interest). Additionally subjective wellbeing effects of unemployment are likely to be related to changes in income [138].

In summary, there seem to be strong conceptual links between urban policies to reduce GHG emissions and wellbeing. It is, however, difficult to quantify this conceptual model, firstly because cities and other health authorities do not often collect WHO-5 or other subjective wellbeing measurements routinely and secondly because a quantitative assessment is often not feasible due to missing information on the nature and the extent of the relationships between urban dimensions and wellbeing. In the next section possible alternatives for wellbeing assessment are described, and other issues to note in the conceptualisation of wellbeing are addressed. GHG reduction interventions at an urban level could have some effect on wellbeing, but as indicated above these could be compensated or counteracted by other factors, such as economic contraction, thus hiding the potential effects of the interventions. Bearing in mind these difficulties, we present below a potential methodological approach for conducting a wellbeing assessment that we have developed for URGENCHE.

4. Quantification of the Theoretical Links between City Conditions and Wellbeing

Our conceptual model makes suggestions about links between wellbeing and GHG reduction emission policies. Quantification of these theoretical links would be useful for cities wishing to carry out a "wellbeing impact assessment" of policies in a similar way to a Health Impact Assessment (HIA).

Literature on linkages between urban conditions (objective wellbeing) which might be affected by the policies in Figure 2, and associated subjective wellbeing which offers an indication to contextualize the conceptual framework and suggestions for further research and discussion, was compiled. As not much evidence was available using the concept of wellbeing itself, allied concepts of mental health and satisfaction were considered in addition to wellbeing, as potential outcomes considered.

Each literature compilation was based on searches of the Web of Knowledge, PubMed and Google Scholar databases. For some of the linkages, very little research was found and Google itself was searched: we supplemented these findings with references known to the authors and for topics with little research, scanning of bibliographies. If an interesting article was found we conducted additional searches for similar articles. If a search did not appear to be generating relevant articles we truncated our evaluation of the papers found. The main searches were started in the traditional way by searching for key words, downloading articles and then searching for relevant papers looking first at the titles then the abstracts and then the full papers. Supplementary searches were made later where only relevant papers were downloaded to databases. The search terms used and references generated from the searches are presented in Table 1. Papers were included in our compilation if they presented statistics on quantitative associations between policy implications (or objective wellbeing) and subjective wellbeing.

Table 1. Searches terms and results.

Policy Area and Search Number	Search Terms *	Total Papers	Papers Providing Quantitative Assessment of Links
BUILDINGS			
1	"((damp/mold/mould) / (thermal comfort/(cold & housing))) & (self-assessed health/mental health/ depression)"	93	9
2	*"(heat stress/air conditioning) & (wellbeing/ depression/ mental health)"*	*NA* **	*1*
TRANSPORT			
1	"(air pollution/noise) & (mental health/depression)"	54	19
2 ***	"(public transport/exercise/physical activity) & (mental health/anxiety/depression)"	568	1 (public transport related)
3	"(commut */transport mode/public transport/active transport) & (social capital/community/social network /volunteer */cultur *)"	51	15
4	*"(accessibility/exclusion) & transport & wellbeing"*	*NA*	*7*
5	*"(green/environment/sustainable) & wellbeing"*	*NA*	*7*
6	*"(affordability/ frugality) & (wellbeing/depression/mental health)"*	*NA*	*5*
INDUSTRY			
1	"(unemployment/employment/job) & (greenhouse gas)"	49	0

Notes: * In some searches these search terms were modified in order to acquire more papers if papers discovered implied other search terms would be beneficial; ** NA (and italic font) indicate "not applicable"—these were supplemental searches where papers were only added to the database if they were found to contain relevant quantitative assessment of links; *** Transport search 2 papers were only considered further if they related to public transport as the relationship between physical activity and mental health was considered established.

Searches for articles on damp and thermal comfort and wellbeing generated reasonable numbers of papers on cold and damp housing. Due to a lack of papers found on uncomfortably hot housing a

supplemental search was conducted but only one paper with quantification of links between too hot housing and wellbeing was found.

Transport policy searches included firstly a search on noise and air pollution and wellbeing. Several studies had attempted quantification of links. Secondly there were searches regarding transport mode and wellbeing, only one reference considered public transport and physical activity, via walking to public transport and only one qualitative reference on public transport and mental health was found. More references were found through searches for active transport and social capital. References on the association between accessibility and mental health were found through supplementary searches but mostly through searches of reference lists.

Transport related GHG reduction policies include encouraging use of alternative transport fuels and in the conceptual model this was posed to affect subjective wellbeing through air pollution alone. Additional supplementary searches were made looking at other consequences of these policies that might affect subjective wellbeing. It was thought that biofuels might lead to subjective wellbeing benefits through adoption of a greener lifestyle and that electric cars might make motoring less affordable.

Although there were 49 references in the energy and employment database, no references presented generalizable quantitative assessments of the relationship between GHG reduction policies and changes in employment industry or power generation.

A selection of findings, with some notes on their potential for quantification of wellbeing effects, is provided in Table 2. We assessed study quality through study design (cross-sectional or longitudinal), sample size, sample location (city, region, or country wide for example) and the statistic that was presented. The choice of statistic affects whether results provided could be used by policy makers to predict the results of implementing a policy on wellbeing in their jurisdiction; ideally we were looking for exposure response functions (ERFs), with the next preference being rate ratios; then odds ratios and the least useful being percentages or proportions.

In traditional HIA, ERFs are used to show that, for example, a reduction in damp in x% homes will lead to a decrease of y% in asthma cases. However, only few ERFs have been estimated as yet between city conditions and wellbeing outcomes (such as between noise and noise annoyance), which makes it very difficult to carry out wellbeing assessments similar to the methods applied for HIA. In addition, there are methodological concerns about those that have been estimated, particularly about the direction of causality [101,139].

In general, the literature search revealed profound weaknesses in existing quantitative approaches to wellbeing measurement. There were conceptual problems with direction of causation, for example does walking increase social capital or are individuals with higher levels of social capital more likely to walk [101,140–149]? Does damp increase depression or are depressed people less likely to deal with damp or more likely to report housing problems? The majority of the studies identified were cross-sectional and thus unable to explore these connections in sufficient depth [150]. It is likely, however, that bi-directional causal models would be needed, involving feedback mechanisms between causes and effects; this would be more challenging than a traditional one-direction causal model [118].

Table 2. Selected examples of wellbeing implications of urban GHG policy implications and potential for quantification of associations *.

Policies	*Implications*	*Objective Wellbeing Aspects Explored*	*Subjective Wellbeing Aspects Explored*	*Notes on Potential for Quantification*
BUILDINGS				
Tightening building envelope & improving insulation	Reduced air flow and reduced heat loss through building envelope	**Mould and damp**	**Depression** **Mental health** **Self-assessed health** **Satisfaction with indoor air quality**	Some evidence of a relationship found [98,150–158] but many studies are cross sectional or based in the UK (particularly the West of Scotland where there is a particular concentration of damp housing and disadvantage). Some odds ratios available.
		Thermal comfort	**Depression** **Mental health** **Residential satisfaction** **Self-assessed health**	Most literature appears to have focused on insufficiently warm housing [152–158] whereas the combination of global warming and increased ventilation may lead to insufficiently cool housing [159]. Some odds ratios available. Differentiation of the effects of cold and damp is difficult.
TRANSPORT				
Tolls & Parking restrictions	Reduce private car use	**Air pollution**	**Depression** **Suicide** **Mental Health**	Fairly consistent findings [160–162] and one Canadian research team has provided relative risks [132,163–165]. However there are many differences by time of year, type of air pollution and gender. Some relative risks available.
		Air pollution	**Annoyance**	ERFs developed for Europe [166,167] but direction of causality could be an issue [139].
		Noise	**Annoyance** **Sleep disturbance** **Mental health** **Depression** **Satisfaction**	Fairly consistent associations [98,118,121,122,168–173]. ERFs developed for annoyance and sleep disturbance [174,175]. Again direction of causality could be an issue [139].

Table 2. *Cont.*

Policies	***Implications***	***Objective Wellbeing Aspects Explored***	***Subjective Wellbeing Aspects Explored***	***Notes on Potential for Quantification***
Tolls & Parking restrictions	Reduce private car use	**Accessibility**	**Mental health**	There is a little, mostly descriptive, research on accessibility and wellbeing mostly from one Australian research team [99,176–181] which is suggestive of an association.
Biofuels	**Leading a green lifestyle**		**Life satisfaction** **Happiness** **Social wellbeing**	A consistent relationship found between leading a green lifestyle and wellbeing but studies have tended to use scales rather than dichotomous outcomes so the search did not find any ratios—generalising from the particular scales used is difficult [182–185].
Electric cars	Cars are less affordable	**Affordability**	**Stress** **Depression** **Happiness**	Studies on affordability and wellbeing are inconclusive [99,182,183,186–188]. Again outcomes tend to be on a continuous scale so ratios were not found.
Promotion of public transport		**Use of public transport**	**Mental health**	A few qualitative & descriptive studies [189] or benefits via extra walking [190].
Cycle paths and foot paths	**More walking and cycling**	**Social capital: informal social networks, community participation, trust, voting**		Studies tend to be cross sectional so difficult to tell the direction of causation [101,140–149]. Odds ratios are available.
		Physical activity levels	**Mental health** **Wellbeing**	There is a vast literature in this area (e.g., [101,191–199]).
INDUSTRY				
Industries encouraged/ discouraged by city	**Change in employment due to cc policies e.g., Power generation**	**Unemployment**	Mental health	One European study has looked at climate change policies and unemployment but the results were not presented in a generalizable manner [113] and other papers are descriptive [200–203].

Note: * Shaded cells with bold font depict relationships which were assessed for quantification.

Secondly there were many concerns about generalizability. Some studies only provided proportions rather than odds ratios or relative risks [138,140,141]; research on some topics has often concentrated on a particular geographical area with a particular culture or weather conditions; additionally many different wellbeing outcomes were measured. Most studies used measures of life satisfaction or mental health, particularly depression rather than subjective wellbeing, often through a plethora of scales rather than a dichotomous measure, so it is not possible to tell the extent to which a score or a change on a particular scale could be generalized to another [167,168,172].

Thirdly it is difficult to distinguish or disentangle concurrent effects, and for example, to establish the extent to which higher levels of depression are related to damp or cold housing, or to generally low socioeconomic status that may be associated with low-quality housing with a higher likelihood for dampness or inadequate thermal performance [93].

Fourthly many studies had not focused on the most relevant aspects, for example most of the literature concentrated on the impact of insufficiently warm housing rather than over warm housing which may be a more pressing issue in settings with reduced ventilation and increasing temperatures. Furthermore some topics, such as the effect of changing power generation source on unemployment, have received little attention within existing research.

In conclusion, there were many relevant studies on wellbeing or wellbeing-related outcomes of urban environmental conditions, but they did not provide the quality of evidence needed for underpinning a wellbeing assessment of specific urban interventions. Few ERF values were found and even when they were identified there were concerns over their validity. Thus new approaches are needed to assess wellbeing effects of policy interventions. Such a new methodology should involve firstly quantification of subjective wellbeing in relation to specific urban conditions to derive risk ratios and allow for wellbeing assessments to be done in the same way as HIAs. If such risk ratios cannot be identified or modelled, other and potentially more crude or basic measures might have to be considered to enable a first, indicative assessment of potential wellbeing impact of urban policies. For such approaches, all data sources providing information on urban conditions and wellbeing could be of interest. Secondly, any new research program should try to take into account that ERFs may be more varying and context related for wellbeing than health outcomes and in the methodology allow for vulnerability across specific groups whose priorities and needs may be completely different and understand and encompass priorities of different stakeholders (both from wellbeing and policy perspectives).

We recommend urban policy-makers take the following steps, based on those underpinning traditional HIA exercises. Firstly baseline levels of subjective wellbeing and city conditions should be determined, perhaps through use or modification of already-existing data and survey methods such as the EQLS [45] which was conducted in 2003, 2007 and 2012 and includes measures of housing conditions, perceived air quality, traffic and greenspace together with measures of subjective wellbeing (WHO-5 (Note that some of the translations of the WHO-5 used in EQLS are different from the translations specified by the developers of WHO-5), happiness and life satisfaction). Alternatively cities with sufficient resources may wish to conduct their own wellbeing survey into which tools targeting subjective wellbeing and life satisfaction should also be embedded (see [33–35] for guidance on various wellbeing measures.)

Secondly estimates are needed regarding the potential effects of policies on urban living conditions (objective wellbeing). The URGENCHE project is developing strategies for estimating such

effects [1,204,205]. This includes, for instance, the relationship between change in traffic flow (via, for example, the implementation of a congestion charge) and air pollution.

Thirdly estimates are needed regarding the relationship between city conditions and subjective wellbeing. For some relationships estimates could perhaps be developed (Table 2) although, given the issues identified above, they should be used with caution. For other wellbeing effects, alternative ways must be found to quantify the effect of a given policy on urban conditions and associated changes in subjective wellbeing at population level; for an example see work by Rehdanz and Maddison [131].

5. Conclusions

In this paper we discuss the theoretical aspects that are to be considered when linking wellbeing to urban policies. In brief, we suggest that urban policies should be evaluated within a broad health perspective that includes wellbeing. Wellbeing assessment requires a consistent conceptual model that can then also enable prioritization of interventions. While we have chosen the WHO-5 scale to describe our conceptual model on environmental influences on wellbeing, other wellbeing approaches may be as reasonable and indeed we need to know the effects of policies on satisfaction with various life domains and overall (as recommended by the WHO Regional Office for Europe Health 2020 policy monitoring framework [49,50]), in addition to developing further understanding of the theoretical and empirical links behind policies' consequences for psychological functioning and hedonic and eudemonic wellbeing.

Here, we have proposed a conceptual model of wellbeing that should make understanding the concept of wellbeing and effects on wellbeing from policies to reduce greenhouse gas emissions easier for local policy makers. Care must be taken when measuring wellbeing to differentiate subjective wellbeing (positive affect) from objective wellbeing (personal, social and environmental conditions that are likely to engender feelings of subjective wellbeing). Dangers of not separating objective and subjective wellbeing may include assumptions by policy makers (and citizens) that high levels of objectively measurable assets are desirable when the literature on subjective wellbeing and socioeconomic status suggests that although disadvantage does reduce subjective wellbeing, affluence does not increase subjective wellbeing [42]. Moreover sustainability issues imply that overconsumption will lead to objective and subjective wellbeing declines for all long term [206].

Climate change policies include buildings, transport, and energy generation interventions and they all theoretically have implications for wellbeing. However there remains a lack of thorough research exploring such interconnections. This lack of attention means that as yet it appears not possible to conduct wellbeing assessments equivalent in rigor to a traditional HIA. However, the compilation of literature reported here did not conform to the stipulations of a systematic review and we recommend that systematic reviews of each of the conceptualized associations are conducted to contribute to future wellbeing assessments of policies. Within these, searches of other databases, such as Cochrane and Psychinfo, should be considered.

Furthermore it is important to acknowledge the local context given the variations found in the wellbeing scores in different settings and cultures, and that wellbeing and the effects of policies are likely to differ by socioeconomic status. The co-existence of environmental exposures and socio-economic factors, known for some agents and some health effects, involve synergistic

interactions; this phenomenon is poorly understood even for physical agents and "hard" health outcomes, so its occurrence in the domain of wellbeing is highly speculative. However, because wellbeing involves perceived health, acceptability of risks and ability to cope with such risks, it can be expected that socio-economic factors such as education may play an important role.

Risk estimates, as well as prevalence differences, can be used to provide some sense of the potential impacts. Depending on the intervention and mechanism, variations of the assessment chain and quantification are possible. These can be used to develop a framework for assessing health and wellbeing effects of policies in order develop priorities for urban policy.

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Author Contributions

Rosemary Hiscock conducted the literature searches, created the conceptual diagram, drafted the majority of the paper and was in charge of the final version. Pierpaolo Mudu wrote the first draft of the sections on spatial and temporal conception of wellbeing in the sustainability section and contributed to the conceptual model. Matthias Braubach contributed to the quantification of the theoretical links between city conditions and wellbeing and the conceptual model. Marco Martuzzi wrote the first draft of the conclusion and was the leader of the project work package on health and wellbeing. Laura Perez contributed to discussions on wellbeing throughout the duration of the project and commented substantively. Clive Sabel was the Principal Investigator of URGENCHE, contributed to the design of the study and promoted the wellbeing aspect of the project throughout. All authors made a substantive contribution to the interpretation of the results and preparation of the manuscript.

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Conflicts of Interest

The authors declare no conflict of interest.

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Abstract: Background
Local strategies to reduce green-house gases (GHG) imply changes of non-climatic exposure patterns.

Objective
To assess the health impacts of locally relevant transport-related climate change policies in Basel, Switzerland.

Methods
We modelled change in mortality and morbidity for the year 2020 based on several locally relevant transport scenarios including all decided transport policies up to 2020, additional realistic and hypothesized traffic reductions, as well as ambitious diffusion levels of electric cars. The scenarios were compared to the reference condition in 2010 assumed as status quo. The changes in non-climatic population exposure included ambient air pollution, physical activity, and noise. As secondary outcome, changes in Disability-Adjusted Life Years (DALYs) were put into perspective with predicted changes of CO_2 emissions and fuel consumption.

Results
Under the scenario that assumed a strict particle emissions standard in diesel cars and all planned transport measures, 3% of premature deaths could be prevented from projected $PM_{2.5}$ exposure reduction. A traffic reduction scenario assuming more active trips provided only minor added health benefits for any of the changes in exposure considered. A hypothetical strong support to electric vehicles diffusion would have the largest health effectiveness given that the energy production in Basel comes from renewable sources.

Conclusion

The planned local transport related GHG emission reduction policies in Basel are sensible for mitigating climate change and improving public health. In this context, the most effective policy remains increasing zero-emission vehicles.

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December 8, 2014

Subject: Manuscript titled "Transport-related measures to mitigate climate change in Basel, Switzerland: a health-effectiveness comparison study"

Dear Editor,

Please find attached our manuscript that assesses the health effectiveness of several locally relevant transport-related climate change policies in Basel, Switzerland.

The results in our manuscript derive from an international research collaboration conducted within the context of the project "Urban Reduction of GHG Emissions in China and Europe (URGENCHE)". Local strategies to reduce green-house gases (GHG) imply changes of non-climatic exposure patterns, but rarely have the health effectiveness of these changes been evaluated. Many mid-size cities all around Europe and other regions in the world are governed by local authorities, thus the findings in our assessment may serve as a model applicable to other regions as well.

We believe our findings that encompass a multidisciplinary approach could be of interest for your readers. We would greatly appreciate the opportunity for this manuscript to go into peer review.

All co-authors qualify for authorship by the criteria of the International Committee of Medical Journal Editors. The content of this manuscript is original, has not been published, either in whole or in part and no similar paper is in press or under review elsewhere. No conflict of interest is reported. All authors have read the manuscript, agree the work is ready for submission, and accept responsibility for the manuscript contents.

We thank you for your consideration of our manuscript.

Sincerely,

Laura Perez
Swiss Tropical and Public Health Institute
Basel, Switzerland

Highlights

Local strategies to reduce green-house gases (GHG) imply changes of non-climatic exposure patterns.

In Basel, Switzerland, we modelled change in mortality and morbidity for the year 2020 based on several locally relevant transport scenarios.

When energy production comes from renewable sources, large diffusion of electric vehicles has the largest health effectiveness ratio.

Many mid-size cities all around Europe and other regions in the world are governed by local authorities, our assessment may serve as a model applicable to other regions as well.

*Manuscript

Transport-related measures to mitigate climate change in Basel, Switzerland: a health-effectiveness comparison study

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Abstract

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1. INTRODUCTION

Urban areas are responsible for up to 70% of the production of carbon dioxide (CO_2) and other greenhouse gas (GHG) emissions causing global warming (UN-Habitat 2011). A set of different policies have been and are being planned, developed and/or implemented in many cities to curb significantly GHG emissions in the future and meet reduction targets. Policies are aimed at different sectors of activities, such as transport, energy production or residential heating.

Many of these climate change policies - sometimes referred to "GHG policies" - imply changes of several non-climatic exposure patterns known to be related to health. Those secondary consequences of GHG policies may be of immediate or mid-term relevance to health whereas the reduction in GHG is a long-term target. Important secondary changes in exposure relate to ambient and indoor air pollutants, redistribution of urban green spaces or mode of transport. These exposures may have direct (positive or negative) effects on health and well-being of the population. The interrelationship between exposure changes, health impacts and climate change policy frameworks was evaluated in a series of papers in 2009 in large European and Asian cities in relation to hypothetical scenarios of policy changes in the household energy, transport, food and agriculture, and electricity generation sectors (Friel and others 2009; Haines and others 2009; Wilkinson and others 2009; Woodcock and others 2009). While these studies have been influential in raising awareness on the large health co-benefits of mitigating climate change, they have been more limited at helping evaluate or design future climate change policies for local policy-makers. Studies that integrated more realistic scenarios of local policy interest have been proven useful (Hankey and others 2012; Rojas-Rueda and others 2011; Rojas-Rueda and others 2012). However, examples in specific local contexts to support local authorities in understanding and forecasting consequences of current and future strategies are still lacking.

Here we present the framework, methodology and results of the health benefits and impacts of locally relevant transport-related climate change policies in the Swiss city of Basel. Basel has been a case study in the 7th European Research Framework program funded project URGENCHE (Urban Reduction of Greenhouse gas Emissions in China and Europe). We focused our analysis on scenarios

related to transport policies. In Basel, the national CO_2 emission reduction targets have been met for several years now and 100% of the electricity is produced by renewable energy (as of 2010). Reducing other GHG emissions (e.g. Nitrogen Oxides and soot particles) still remains a need related principally to the transport sector. Despite a long history of environmental sustainability policies and synergetic collaboration across relevant city departments, health aspects in Basel have never been integrated in the evaluations of the effectiveness of transport policies.

2. METHODS

We modelled differences in health impacts between current conditions (2010 year of reference), assumed as status quo until year 2020, and the year 2020 under different GHG transport-related policy scenarios of relevance for Basel. The target population of our analysis included only the residents of the canton of Basel-Stadt, the administrative unit where Basel city is located. Thus we only account for impacts of policies that may contribute to local changes. The methodological steps followed for the health impact assessment (HIA) include selection of relevant scenarios for the city, selection of outcomes and concentration-response functions (CFRs) available from current revision of the epidemiological literature and HIA practices, modelling changes in population exposure under the different selected scenarios and combining all the above for calculating the health impact resulting from predicted changes in exposures.

2.1 Choice of pathways of exposure and scenarios

Transport policies act on urban development and mobility aspects and imply changes in CO_2 emissions and exposure of the population to short-lived air pollutants such particles or nitrous oxide (NO_2), as well as primary pollutants near to transport sources, such as soot. These policies also imply changes in noise exposure or physical activity patterns and in general of time-activity that may be linked to accidents and further exposure to air pollution. Changes in mobility patterns are expected to influence fuel usage within the city limits with additional consequences for CO_2 emissions. For Basel, the pathway of exposure evaluated as a result of these changes include predicting changes in CO_2 emissions, regional pollution (represented by particulate matter up to 2.5 micrometer in diameter [$PM_{2.5}$]), near-road traffic-related pollution (represented by elemental carbon [EC]), noise and physical activity patterns.

We retained four scenarios of transport changes compared to the reference scenario that assumes that all conditions in 2010 remain as they are until the target year 2020, except for population size:

(1) a scenario that includes all transport-related measures that are decided by the local government to be developed up to 2020. This scenario relies on the assumption that those policies are being currently implemented and likely to be maintained unless jeopardized by future political processes and decisions. We refer to this scenario as the “Decided Policies” (DP);

(2) the so-called “Z9 scenario” developed by the city that accounts for additional local transport measures beyond DP to further reduce traffic by 4% on inner roads. This local scenario complies with a successful citizen referendum requiring further traffic reduction. It includes a series of traffic measures targeted at channelling traffic along main avenues, reduce traffic and moderate speed in residential areas not contemplated in DP. This scenario also assumes a local shift of car trips to active transport (walking and cycling);

(3) A hypothetical scenario named “p10” assuming a 10% reduction of traffic on inner roads as compared to DP; in p10, all other measures of Z9 are also assumed to be implemented, thus the only difference between Z9 and p10 is the 10% instead of only 4% reduction in traffic; and

(4) the p50 scenario expanding p10 with the assumption that 50% of the private car fleet would be based on electric vehicles (p50).

While DP and Z9 were locally developed by the city authorities, p10 and p50 are default scenarios that the URGENCHE project team agreed upon for all cities included in the project.

2.2 Selection of outcomes and concentration-response functions

Table 1 summarizes exposure and outcomes retained in our analyses. We used $PM_{2.5}$ as the indicator of long-term exposure to air pollution. $PM_{2.5}$ has been associated with total, cardio-respiratory and lung cancer mortality and shortened life expectancy (WHO 2013a; WHO 2013b). Our core analysis includes the estimation of long-term exposure to $PM_{2.5}$ on all-cause natural mortality and cause-specific mortality (lung cancer and cardiovascular diseases) in sensitivity analyses (Hamra and others 2014; Hoek and others 2012). In recent years, epidemiological studies have shown that local traffic emissions may be independently related to the long-term health effects of regional air pollution (HEI 2009). While these local primary emissions contribute to only a small fraction of the total mass of $PM_{2.5}$, their distribution depends on the distance to source. This is less the case of $PM_{2.5}$ that is more

homogeneously distributed. Thus, studies using $PM_{2.5}$ may not fully capture the effects of near-road traffic-related particles. It has been suggested that indicators of near source combustion particles such as black carbon characterized by EC could be used in addition to $PM_{2.5}$ to evaluate the effects of local traffic sources (Keuken and others 2012a). In order to avoid double counting, the effects due to exposure to EC should not be added to the effects related to exposure to $PM_{2.5}$ (WHO 2013a). We conducted a sensitivity analysis using EC as pollutant indicator to compare results with $PM_{2.5}$ on mortality (Janssen and others 2011). There are also short-term effects of $PM_{2.5}$, such as increases in hospitalizations for cardiovascular and respiratory causes, and other minor ailments such as restricted activities days (RADs) evaluated in the sensitivity analysis. We used CRFs and baseline data as recently recommended in WHO reviews (Hurley F. and others 2005; WHO 2013a; WHO 2013b).

There is growing evidence of effect of noise on several health outcomes such as sleep disturbance, annoyance, hypertension, myocardial infarction, and tinnitus (EEA 2010). Of special relevance is that noise may have independent effects from air pollution on the cardiovascular system. We evaluated the health effects of noise on mortality by coronary heart diseases using a recent newly developed risk function (Babisch 2014). Annoyance and sleep disturbance have been traditionally recommended to be included in HIA from noise (EEA 2010). It is yet unclear how these outcomes relate to the more distal cardiovascular risk, as individuals more disturbed may also be more susceptible. We used these outcomes as a measure of independent impact on well-being (Miedema and Oudshoorn 2001; Miedema and others 2003).

The relative risk for an association between physical activity and health effects have been reported in recent reviews (Woodcock and others 2011). We retained for this analysis the outcome and risk function developed and implemented in the WHO's Health economic assessment tool (HEAT) for cycling and walking that estimates changes in all-cause premature deaths given changes in walking and cycling (WHO 2014). Specifically we used a linear risk function for cycling that estimates a reduction in mortality of 10% for regular commuter cyclists compared to non-cycling commuters. A regular commuter cyclist was defined as a commuter aged between 20 and 64 who cycles at least 100

minutes per week for 52 weeks per year. For walking, HEAT proposed a risk reduction of 11% for walkers compared to non-walkers. Walkers are defined as individuals aged between 20 and 74 years that walk at least 168 minutes per week for 52 weeks per year. While the CRFs used in this analysis were developed on a specific range of ages, for consistency our analysis is applied to ages above 30 years for all outcomes except RADs which was applied to the population 18-64 years and walking and cycling which was applied to those aged 15-74 years (Table 1).

2.3 Population exposure modelling

Annual $PM_{2.5}$ and EC concentration maps for 2010 and 2020 were developed for the different policy scenarios. We obtained annual regional and urban background concentrations and road traffic contributions by combining different models. The regional annual background concentrations of $PM_{2.5}$ and EC in 2010 and in 2020 were modelled by a regional scale models with a urban and adding 1 x 1 km urban traffic emissions to the regional background. The road traffic-related $PM_{2.5}$ and EC concentrations were further developed using a 'street canyon' model applied to urban streets, and a 'line-source' model to motorways. Detailed methods have been described elsewhere (Keuken and others 2014) and are summarized in Supplemental Materials. We used traffic models developed in by Basel city for DP and Z9 as input in the models. We developed additional traffic maps for p10 and p50 according to the assumed scenarios.

We used noise exposure levels for DP and Z9 developed by the city of Basel (see Supplemental Materials). Models of change in traffic noise given replacement of fleets with electric cars are poorly developed on the scale of cities like Basel, thus, the adoption of the above models for p10 and p50 come with major uncertainties. Thus, we adopted an alternative p50-scenario (p50*) for Basel that assumed a 2dB Lden reduction in the whole city if levels were above background compared to 2010. This scenario matched with local predictions developed previously in the city within the context of a pilot study on energy savings in Basel that assumed 80% of electric vehicles year 2030 (KBS 2011).

The health impact evaluation for the change in physical activity exposure was conducted only for Z9 for which an additional total 27,000 walking and cycling within-city trips per day has been estimated by the city (BKB 2008). No further modelling of the distribution of these new trips by transport mode was conducted at the city level. Thus, to estimate the percentage increase in trips per active transport mode for Z9 compared to the reference scenario, we assumed that the current share of cycling and walking trips as observed in the Swiss micro-census – a population-based survey - for the Canton of Basel-Stadt in 2010 would remain the same (SFOE 2010). Active trips at reference level were calculated using the ratio of total trips and active trips seen in the micro-census sample. Total active trips for each active transport mode for the Z9 scenario were afterwards estimated as the sum of the trips at the reference level and the 27’000 additional estimated active trips by the city.

For air pollution and noise, population-weighted concentrations were calculated for each scenario superimposing the spatially resolved maps with population data given at the building address level. To evaluate the variability of residential exposure to near-traffic sources, we further calculated population-weighted concentration by type of residential zone namely population living within 100 m of a motorway (Zone 1), people living 25 m from a street with more than 10,000 vehicles per day (street canyon) (Zone 2), and population living in zones other than zone 1 and 2, assumed as next to inner city roads (Zone 3). Population for 2010 was obtained from the Basel statistics department. (Statistisches Amt des Kantons Basel-Stadt). Figures of population growth for Basel provided by the Federal Office of Statistics (BfS) were used to calculate Basel population in 2020 at building entrance, by sex and 5-year age groups.

2.4 Calculation of health impacts

The primary impact metric of this analysis is the difference per year in number of premature deaths and morbidity cases due to each policy scenario compared to the reference level using standard population attribution fraction and life tables methodologies (Miller and Hurley 2006; Perez and Künzli 2009). We additionally developed a comparative risk assessment by using as a secondary

metric life-long changes in Disability Adjusted Life Years (DALY) expressed per 1,000 inhabitants. For each pathway of impact, DALYs were calculated as the sum of YLL gained (or lost) due to deaths prevented (or brought forward) and additional Years of Life lost due to Disability (YLD) from outcomes related to well-being aspects (RAD for air pollution exposure and annoyance and sleep disturbance for noise). For RAD, we estimated attributable cases using rates proposed in the WHO review given lack of local data (WHO 2013a). YLD were then estimated as the total number of attributable RADs due to change in exposure multiplied by a disability weight of 0.099 per year (Hänninen and Knol 2011). We used previous non-linear function to obtain the percentage change of people highly annoyed or with disturbed sleep given the average residential level of noise in the target population for each scenario and the reference level and calculated differences (Miedema and Oudshoorn 2001; Miedema and others 2003). YLD were then estimated using a disability weight of 0.02 per year for both outcomes as recommend in the WHO noise health impact guidelines (EEA 2010). There was no discounting in years or age.

Finally, the results of the comparative analysis were compared with changes in CO_2 emission reduction and fuel consumption. CO_2 emission factors for CO_2 and fuel consumption by vehicle type were combined with traffic intensities by vehicle and road type in Basel to obtain yearly CO_2 emissions and fuel consumption for each scenario.

3. RESULTS

3.1 Population and baseline mortality data

Table 2 presents a summary of the population and baseline health data used for the impact calculations. The population of Basel in 2010 accounts for 191,257 inhabitants (51% women). Over 68% of the total population is between the ages of 15 and 64 years. About 5% of the total population is

above 80 years old (7.9% for women). Most of the population lives away from highways and major roads (90%). In 2020, an increase in population of about 2.5% is expected (to 196,066 inhabitants).

There were a total of 1884 natural deaths registered in 2010 in Basel above the age of 30 years, of which 57% were women. Most of these deaths occurred in people above 80 years old (67% of all deaths), although there was a marked difference between genders (55% of deaths due to natural cause in men above 80 years versus 79% for women).

3.2 Predicted exposure changes

The predicted change of $PM_{2.5}$ and EC under the DP scenario was estimated to decrease by 38% and 66% respectively as compared to the reference (Table 3). However, the models estimate the additional reduction of $PM_{2.5}$ and EC for the scenarios beyond DP to be very small. There was considerably less change in average Lden and Lnight exposure for any of the scenarios considered (<2%) with DP and Z9, resulting in a slight increase in the population-weighted mean exposure compared to reference. The change in air pollution exposure did not depend on the residential zone while there was a larger difference in changes for noise indicators for inhabitants of residential zone 1 (increase).

Within Basel, in 2010, 15% and 41% respectively of all trips are done by cycling and by walking averaging 16 and 48 minutes per day, respectively. We estimated that the Z9 scenario could result in about 7 percent point increase in the predicted number of cycling and walking trips compared to reference level 2020 or 7,222 (calculated as 108,049-100,827 trips) and 19,778 (calculated as 291,068-271,290) new daily cycling and walking trips, respectively (Table 2). This would represent an increase in the share of active trips compared to total trips of about only 1%. We found a slightly larger increase for women compared to men.

3.3 Health impacts

Postponed deaths and prevented RADs are reported in Table 4. We estimated that per year there could be a reduction in natural deaths by up to 3% (65 deaths) if $PM_{2.5}$ exposure levels were further reduced to DP-scenario levels. Given the minor additional reduction of exposure in the other scenarios, those contributed only moderately to a further reduction of impacts. Cause-specific analyses show that a large contribution of these attributable deaths are related to cardiovascular diseases (CVD) (50 of the 65 preventable deaths). Using EC as an indicator of near-road traffic pollution increases the preventable deaths to 6% (115 cases) per year for DP. We estimated that per year there are more than 2.3 million RADs in Basel and that 2.0% of those could be reduced if DP levels were achieved for $PM_{2.5}$.

In general, the benefits or impact of noise for mortality and annoyance and sleep disturbance were very limited given the small changes in the population weighted mean noise levels predicted by the scenarios. We found that 1% - corresponding to less than 7 CVD deaths - are expected to be postponed due to changes in noise exposure expected in the p50 scenario.

We estimated that about 0.03% premature deaths (<1 natural death in commuters) per year could be prevented if there was 1% more walking and cycling in Basel. The negative impact due to additional air pollution exposure while commuting was negligible (results not shown).

3.4 Comparative risk assessment

The comparative assessment to contribution of the reduction in DALYs due to changes in exposure and scenarios shows that near-road traffic reduction remains the largest benefit for DP (-3.8 DALYs per 1,000 population) with other scenarios bringing altogether only marginal benefits beyond DP (Table 5). The comparative analysis however shows that noise reduction from electro mobility contributes to reducing some of the well-being impacts. Representing a measure of the health

effectiveness of the scenarios compared to reference level, the ratio between the differences in CO_2 emissions versus the differences in health benefits is the largest for the p50 scenario.

4. DISCUSSION

Our study evaluated the impacts on health of realistic local GHG reduction-related policy scenarios in an environmentally friendly and climate change sensitive middle-sized urban city in Europe. Many mid-size cities all around Europe and other regions in the world are governed by local authorities, thus the assessment may serve as a model applicable to other regions as well. Our study further puts into perspective potential health changes due to local realistic measures against changes from more ambitious hypothesized interventions such as large diffusion of electric cars that are less likely to be implemented in the near future although reflecting targets that are discussed in Basel and elsewhere.

Our results show that DP, including a range of already planned transport measures, will bring relatively large air pollution-related health benefits. This is principally due to the anticipated reduction of tail-pipe particle emissions from the new fleet of Euro 5/6 diesel engine vehicles expected to be in place by 2020 (Keuken and others 2012b; Kousoulidou and others 2008). Thus, we show for the first time that DP is also of major health relevance and authorities should remain very focused on its full implementation. If the adoption of DP gets delayed or diffused – e.g. through new political initiatives against some of the DP measures - our beneficial DP results may be an overestimation of the true changes. Our study showed that further reductions in EC exposure are of substantial health relevance and clearly more sensitive to local policies than $PM_{2.5}$ because population exposure to EC (as an indicator for soot emissions) is strongly driven by local traffic-related emissions. Most air pollution impact assessments still rely solely on $PM_{2.5}$ as long-term indicator of health. Using $PM_{2.5}$ alone may be underestimating the benefits of local transport measures and, thus, underestimating the contribution of local policies to improvement in public health. Outdoor air pollution and diesel particles that mostly contribute to EC composition are now recognized as carcinogenic to humans (Hamra and others 2014; Raaschou-Nielsen and others 2013). For diesel particles, a level of 0.1 ug/m^3 of EC exposure over the lifetime to meet the acceptable risk of 1 in a million- of developing one additional cancer due to

exposure (SoCAB 2008). Current levels and future scenarios resulted in exposure levels of EC well above this threshold showing that there is still potential for large gains beyond reducing regional air pollutants such as $PM_{2.5}$, often the only target of most air pollution and GHG reduction policies. Our results show in any case that zero emission vehicles will have very large benefits in many urban areas. Thus, the results support efforts to implement air quality standards that regulate the local traffic-related pollutants in addition to those capturing the background levels (such as $PM_{2.5}$). In spring 2014, The Swiss Federal Commission for Air Hygiene (FCAH) recommended to the government the inclusion of a binding 80% reduction of EC at all locations in within the next 10 years in its air quality law (CFHA 2014). Our study indicates that DP and the other scenarios will reduce EC exposure to 30-35% of current levels by 2020. Assuming continued and additional efforts to reduce traffic-related emissions after 2020, it appears rather plausible to curb the EC concentrations down to 20% of current levels by 2025. Assuming linear interpolation, compared to our reference point, the FCAH scenario would result in 59 prevented deaths, or some 17% higher benefits as compared to the best scenario considered in URGENCHE (p50). FCAH also proposed the adoption of the WHO guideline value for the $PM_{2.5}$ annual mean, namely to set 10 $\mu g/m^3$ $PM_{2.5}$ as a national air quality standard. According to our estimates, the DP scenario, if fully implemented and effective, is likely to achieve and go over this standard for the citizens of Basel by 2020.

Several studies have shown that the shift from private vehicle to active transport is a key intervention for improving public health, both physically and psychologically (Hankey and others 2012; Maizlish and others 2013; Rojas-Rueda and others 2011; Rojas-Rueda and others 2012; Woodcock and others 2013; Woodcock and others 2014). In Basel, the baseline level of active transport is already high so predicted growth in numbers of new active travellers under the Z9 scenario was rather small, with benefits of these measures remaining modest compared to past studies. For example, in Barcelona, it was estimated that 12 deaths per year could be postponed under a scenario assuming that 25,425 individuals replace car use with bicycle riding. In a study in England and Wales, a reduction of premature deaths between 3% and 9% was estimated assuming increased levels of walking and cycling could reach up to 37% (Woodcock and others 2013). In London a hypothetical increase in

active travelling by up to 5 times the travel times observed in 2020, prevented 528 premature deaths projected for year 2030 (Woodcock and others 2009). From a public health, traffic management, and urban planning perspective, such strategies are very appealing not only for Basel but indeed on the global scale. This is particularly true for Chinese cities – like those participating in URGENCHE – where policy makers should take all efforts to keep the traditionally very common use of bikes very high on the urban planning agenda to not continue the unfortunate shift from bikes to cars.

As consequence of further increases in bicycle use, and in particular a substantial increase in the use of electric bicycles, there is in Switzerland and in Basel, an upward trend in accidents and injuries due to bicycle riding (BFPA 2014). Some studies have pointed to the potential increase in traffic injuries as a negative impact of these measures; in the San Francisco Bay, an increase in active commuting time was estimated to contribute an increase of 39% of the injury burden (Maizlish and others 2013). Given the uncertainty in Basel relating to our transport shift scenario we abstained from further analysis. Active transport policies should be accompanied by additional strategies for the safe use of bicycles. Though the city of Basel has adopted some of these measures, other regions such as the city of Copenhagen have promoted bicycle use with far more rigorous and coherent concepts (Fraser and Lock 2011).

Our results show that more ambitious hypothesized scenarios considering large penetration of electric cars in the city in the year 2020 did not contribute considerably to increased health benefits from noise reduction and that an increase in population exposure to noise and related negative health impacts is even predicted under the DP scenarios. It is estimated that at 50 km/h an electric car emits ~ 1 dB less than a combustion fuelled car and at 30 km/h this is ~ 2 dB (HEIMTSA 2011). Our evaluation suggests that the reduction of noise in inner roads even if the fleet of private cars is mostly electric would not compensate for the increase in exposure to noise from more cars in high speed roads, expected to occur in most European cities. Thus, instead of accepting this increase as “a default”, policy makers need to be innovative to abate these trends. Moreover, our scenario assumed a conservative mix of electric and combustion-driven vehicles, i.e. every second car remains noisy.

Policies that would restrict access to some densely populated areas of the city to electric cars only would lead to stronger noise reductions in those areas than considered in our scenario. Despite these moderate benefit compared to air pollution reduction, our study shows that this measure has the largest health effectiveness ratio when the energy production is principally from renewable energy like in Basel as confirmed in other settings (Buekers and others 2014). However one should be aware that the realistic level of diffusion of electric cars in the period study is rather small. In Europe, only 7% market share for electric vehicles is expected in 2020 (EC 2011).

Our study presents several limitations. Our results are represented only by a central estimate, i.e. uncertainties are not specifically quantified. Impact studies like ours remain projections based on a large number of assumptions behind each step of the calculation. We are not able to validate each assumption but only discuss potential impact for our conclusions. We took an approach to minimize double counting of benefits and impacts, but altogether the benefits may represent an underestimation. For example we did not account for all impacts of changes in air pollution or physical activity in health and we did not consider in depth well-being aspects beyond annoyance and sleep disturbance. Another limitation relates to the fact that we do not know how health in the population will be modified given other contextual changes that may occur at local level, thus, our scenarios assume that all other health-relevant aspects remain stable. Given the short time frame of our analysis, we can only speculate that current conditions will apply to the future. We focused our analysis on transport-related policies given the interest of local policy-makers. Moreover, the benefits of the most effective scenario – DP - will only materialize if those measures do not become subject to new political initiatives to stall their implementation. Given the Basel context, we expect that any additional health benefits from a different sector of activity would only be marginal, unless far more substantial and possibly visionary scenarios were considered. Benefits and impacts of other exposure changes (i.e. heat, green spaces) over a longer time-frame were beyond the scope of this study. The distribution of exposure and impacts in relation to socio-economic status is also rather marked in Basel. Our analysis clearly showed differences by sex originating both due to difference in baseline health but also in exposures as is the case for physical activity and active commuting. We did not stratify further our results given

the small population but these aspects should be better considered when developing future measures. Useful lessons in this regard could be drawn from joint analysis of the results obtained in other, larger cities involved in URGENCHE, such as Thessaloniki in Greece. Such a comparison will be the topic of future work on this theme.

5. CONCLUSIONS

The planned local transport-related GHG emission reduction policies in Basel are sensible for mitigating climate change and improving public health. In this context, the most effective policy remains increasing zero-emission vehicles. The benefits of such policies can be remarkable even in cities like Basel where environmental pollution is at only moderate levels due to long-term investments in policies to protect the environment and, thus, people's health.

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Table(s)

Table 1: Selected health indicators, relative risks and level of analysis*

Health endpoint and population applied to	Outcome definition (baseline incidence)	Quantitative association of exposure with health endpoint (RR)	Source of RR	Core analysis	Sensitivity analysis	Comparative Risk Analysis
Due to PM $_{2.5}$						
All-caused mortality excl. accidents (30 years and older)	A00-R99	1.062 (95% CI: 1.040 - 1.083) per 10 $\mu g/m^3$	Meta-analysis, Hoek et al. (2013)(Hoek et al. 2013)	X		
Mortality due to cardiovascular disease (30 years and older)	I00-I99	1.15 (95% CI 1.04; 1.27) per 10 $\mu g/m^3$	Meta-analysis, Hoek et al. (2013)(Hoek et al. 2013)		X	
Lung cancer related mortality (30 years and older)	C33	1.09 (95% CI 1.04; 1.14) per 10 $\mu g/m^3$	Meta-analysis, Hamra et al, 2014 (Hamra et al. 2014)		X	
RADs, Restricted activity days (18-64 years)	1,900,000 RADs per 100,000	4.75% (95% CI: 4.17 - 5.33) per 10 $\mu g/m^3$	Hurley et al. (2005)(Hurley F et al. 2005)			X
Due to EC						
All-cause mortality excl. accidents (30 years and older)	A00-R99	1.06 (95% CI: 1.04, 1.09) per 1 $\mu g/ m^3$	Janssen et al. (2011)(Janssen et al. 2011)		X	X
Due to Noise						
Cardiovascular mortality (30 years and older)	I00-I99	1.08 (95% CI: 1.04, 1.13) Lden per 10 dB	Babish (2014)(Babisch 2014)	X		X
High Annoyance** (30 years and older)	-	$\%HA = 9.868*10^{-4} (L_{den}-42)^3 - 1.436*10^{-2} (L_{den}-42)^2 + 0.5118 (L_{den}-42)$	Miedema and Oudhoorn (2001)(Miedema and Oudshoorn 2001)			X
Highly Sleep disturbed** (30 years and older)	-	$\%HSD = 20.8 – 1.05 * L_{night} + 0.01486 * L_{night}^2$	Miedema and Oudshoorn (2003) (HME. et al. 2003)			X
Due to cycling and walking						
Cycling All cause mortality (15-74 years)	A00-R99	0.90 (95% CI: 0.87 - 0.94) per 100 minutes/week	Meta-analysis, WHO, 2014(WHO 2014)	X		X
Walking All cause mortality (15-74 years)	A00-R99	0.89 (95% CI: 0.83 - 0.96) per 168 minutes/week	Meta-analysis, WHO, 2014 (WHO 2014)	X		X

*Air pollution and noise health impacts were conducted at the building level, for which population density by age and sex was available, and aggregated over the study area. Health impacts of cycling and walking were conducted aggregated over the study area.

Noise: Percentage and number of adults annoyed and highly annoyed, indoor, in 1 year. Percentage and number of adults sleep disturbed and highly sleep disturbed, indoor, in 1 year.

Table 2. Baseline population description for Basel residents, year 2010

Description	Ages	Male	Female	Total
Total Population	All ages	94214	97044	191257
Population distribution (% total population)	0-1 yrs	1.1%	0.9%	1.0%
	2-14 yrs	15.5%	13.0%	14.2%
	15-64 yrs	71.2%	65.3%	68.2%
	65-80 yrs	9.8%	12.9%	11.3%
	≥80yrs	2.4%	7.9%	5.2%
Residential location (% total population)				
within 100m of an urban motorway (Zone1)	--	--	--	2.5%
living 25m from a street with more than 10'000 vehicles per day (street canyon, Zone 2)	--	--	--	7.5%
for the rest of the population, inner roads (Zone 3).)	--	--	--	90%
Cause of deaths (%≥80yrs)				
All natural cause (A00-R99)	≥30 yrs	813 (55%)	1071 (79%)	1884 (67%)
Cardiovascular (I00-I99)	≥30 yrs	256 (68%)	384 (90%)	640 (79%)
Lung Cancer (C34)	≥30 yrs	57 (26%)	26 (31%)	83 (29%)
Well-being				
Restricted activity days	20-64yrs	--	--	2.8mio
Highly annoyed by noise (%total population)	≥30 yrs	--	--	6.0%
Highly sleep disturbed by sleep (%total population)	≥30 yrs	--	--	4.6%
Active travellers	15-74 yrs			
cycling trips, %total trips		15%	15%	15%
walking trips %total trips		35%	45%	41%
Average daily duration of cycling trips (minutes)		15	17	16
Average daily duration of walking trips (minutes)		40	57	48
Estimated number of cycling trips (2020)		50163	50664	100827
Estimated number of walking trips (2020)		116379	154911	271290
Estimated number of cycling trips (Z9)		53288	54761	108049
Estimated number of walking trips (Z9)		123629	167439	291068

Table 3. Population weighted concentrations (% change compared to reference), given by pollutant, scenario and residential area

		All city		Zone 1 near mortoways		Zone 2 near major roads		Zone 3 inner roads	
PM2.5 (µg/m^3)	Reference	15.05	--	15.36	--	15.65	--	14.99	--
	DP	9.40	(-38%)	9.51	(-38%)	9.56	(-39%)	9.39	(-37%)
	Z9	9.39	(-38%)	9.49	(-38%)	9.52	(-39%)	9.38	(-37%)
	p10	9.39	(-38%)	9.49	(-38%)	9.53	(-40%)	9.38	(-38%)
	p50	9.36	(-38%)	9.45	(-38%)	9.38	(-39%)	9.36	(-37%)
EC (µg/m^3)	Reference	1.58	--	1.80	--	2.00	--	1.54	--
	DP	0.53	(-66%)	0.61	(-66%)	0.64	(-68%)	0.52	(-66%)
	Z9	0.52	(-67%)	0.59	(-67%)	0.61	(-68%)	0.51	(-67%)
	p10	0.52	(-68%)	0.60	(-69%)	0.62	(-74%)	0.51	(-67%)
	p50	0.50	(-67%)	0.57	(-67%)	0.52	(-69%)	0.50	(-67%)
Lden (dBA)	Reference	50.82	--	59.71	--	66.78	--	49.24	--
	DP	52.12	(0.4%)	59.69	(1.7%)	65.36	(0.7%)	50.81	(0.2%)
	Z9	52.01	(-0.3%)	60.72	(1.3%)	66.52	(0.3%)	50.56	(-0.4%)
	p10	--	--	--	--	--	--	--	--
	p50*	49.43	(-0.4%)	57.85	(0.7%)	64.83	(-0.7%)	47.92	(-0.2%)
Lnight (dBA)	Reference	41.01	--	50.74	--	58.02	--	39.33	--
	DP	41.83	(0.5%)	50.40	(2.0%)	56.15	(0.9%)	40.40	(0.2%)
	Z9	41.80	(-0.4%)	51.51	(1.6%)	57.38	(0.4%)	40.24	(-0.4%)
	p10	--	--	--	--	--	--	--	--
	p50*	39.73	(-0.4%)	48.90	(0.8%)	56.07	(-0.8%)	38.12	(-0.3%)

Table 4.Number of deaths or cases prevented or brought forward (% of expected 2020 death for age group considered) by pollutant, outcome, and scenario

Exposure	Outcome (number deaths in age group considered)	Scenario	Male		Female		Total	
$PM_{2.5}$	Natural deaths	DP	-28	(-3.3%)	-37	(-3.3%)	-65	(-3.3%)
	(age≥30, 1951)	Z9	-28	(-3.4%)	-38	(-3.4%)	-66	(-3.4%)
		p10	-28	(-3.3%)	-37	(-3.3%)	-65	(-3.3%)
		p50	-28	(-3.4%)	-38	(-3.4%)	-66	(-3.4%)
	Cardiovascular deaths	DP	-20	(-7.6%)	-31	(-7.6%)	-50	(-7.6%)
	(age≥30, 665)	Z9	-20	(-7.6%)	-31	(-7.6%)	-51	(-7.6%)
		p10	-20	(-7.6%)	-31	(-7.6%)	-51	(-7.6%)
		p50	-20	(-7.6%)	-20	(-5.0%)	-40	(-6.0%)
	Lung cancer deaths	DP	-3	(-4.7%)	-1	(-4.7%)	-4	(-4.7%)
	(age≥30, 85)	Z9	-3	(-4.8%)	-1	(-4.8%)	-4	(-4.8%)
		p10	-3	(-4.8%)	-1	(-4.8%)	-4	(-4.8%)
		p50	-3	(-4.8%)	-1	(-4.8%)	-4	(-4.8%)
	Restricted activity days	DP	-29155	(-2.3%)	-27728	(-1.8%)	-56883	(-2.0%)
	(age20-64yrs, 2.8mio)	Z9	-29354	(-2.3%)	-27917	(-1.8%)	-57271	(-2.0%)
		p10	-29211	(-2.3%)	-27781	(-1.8%)	-56993	(-2.0%)
		p50	-29354	(-2.3%)	-27917	(-1.8%)	-57271	(-2.0%)
EC	Natural deaths	DP	-49.16	(-5.9%)	-66.23	(-5.9%)	-115.39	(-5.9%)
	(age≥30, 1951)	Z9	-50.39	(-6.1%)	-67.89	(-6.1%)	-118.28	(-6.1%)
		p10	-49.51	(-6.0%)	-66.70	(-6.0%)	-116.22	(-6.0%)
		p50	-50.39	(-6.1%)	-67.89	(-6.1%)	-118.28	(-6.1%)
Noise	Cardiovascular deaths	DP	2.74	(1.0%)	4.19	(1.0%)	6.93	(1.0%)
	(age≥30, 665)	Z9	-2.72	(-1.0%)	-4.17	(-1.0%)	-6.90	(-1.0%)
		p10	--	--	--	--	--	--
		p50*	-2.72	(-1.0%)	-4.17	(-1.0%)	-6.90	(-1.0%)
	High annoyance	DP	5	(0.1%)	6	(0.1%)	11	(0.1%)
	(age≥30, 7775)	Z9	24	(0.6%)	29	(1%)	53	(1%)
		p10	--	--	--	--	--	--
		p50*	-121	(-3.3%)	-148	(-4%)	-269	(-3%)
	High sleep disturbance	DP	-57.61	(-1.6%)	-70.71	(-1.7%)	-128.32	(-1.7%)
	(age≥30, 5961)	Z9	-43.04	(-1.2%)	-52.83	(-1.3%)	-95.87	(-1.2%)
		p10	--	--	--	--	--	--
		p50*	-40.34	(-1.1%)	-49.52	(-1.2%)	-89.86	(-1.2%)
Physical Activity	Natural deaths (age15-74yrs, 205)	Z9	-<1	(-0.08%)	-<1	(-0.09%)	-<1	(-0.17%)
		Z9*	-<1	(-0.08%)	-<1	(-0.09%)	-<1	(-0.17%)

Percent change in deaths represent changes compared to expected death in 2020 in age group outcome applies to. P50*: Considers 2dB reduction in whole city; Z9*: corrected by negative effects of more air pollution breathing during cycling and walking commuting.

Table 5. Comparative risk assessment expressed as reduced or increased Disability-Life Years (DALY) per 1,000 population with CO_2 emission and fuel consumption changes compared to reference

	DP	Z9	p50
Air pollution			
Mortality	-3.7	-3.8	-3.8
Restricted activity days	-0.08	-0.08	-0.08
sub-total	-3.8	-3.9	-3.9
Noise			
Mortality	0.10	-0.10	-0.10
High annoyed	0.02	0.09	-0.46
High sleep disturbance	-0.22	-0.16	-0.15
sub-total	-0.10	-0.17	-0.71
Physical activity, mortality	--	-0.06	--
Total DALY	-3.7	-4.0	-4.0
CO_2 emission	6%	-1%	-20%
Fuel consumption	9%	--	-15%
Ratio ΔCO2/ ΔDALY	-3	--	9

Supplementary Information
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Article

Development of a quantitative methodology to assess the impacts of urban transport interventions and related noise on wellbeing

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Abstract: Wellbeing impact assessments of urban interventions are a difficult challenge, as there is no agreed methodology and scarce evidence on the relationship between environmental conditions and wellbeing. The EU project "Urban Reduction of Greenhouse Gas Emissions in China and Europe" (URGENCHE) explored a methodological approach to assess wellbeing impacts of transport-related interventions in three European cities (Basel, Rotterdam and Thessaloniki). The approach connects modeled noise reduction effects of the interventions with survey data indicating noise-wellbeing associations.

Local noise models showed a reduction of high traffic noise levels in all cities as a result of the urban interventions. Survey data indicated that perception of high traffic noise levels was associated with lower probability of wellbeing. Connecting the local noise exposure profiles with the noise-wellbeing associations suggests that the urban transport interventions may have a marginal but positive effect on wellbeing.

The paper shows that wellbeing changes as a result of environmental interventions can be estimated. However, due to a lack of suitable data and methods for wellbeing impact assessment, several assumptions have to be made and the results are relatively indicative.

Keywords: Urban policies, climate change, mitigation, greenhouse gas, transport, noise, wellbeing, impact assessment.

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1. Introduction

1.1. Context and project objectives

Cities produce approximately 75% of the global carbon emissions (1) and therefore are in a unique position to facilitate effective reductions of greenhouse gas emissions. The EU project “Urban Reduction of Greenhouse Gas Emissions in China and Europe” (URGENCHE) aimed to quantify health and wellbeing co-benefits of urban policies targeted at greenhouse gas (GHG) emission reduction as required by the Kyoto Protocol commitment for a 20% GHG reduction by 2020 (2,3). The urban policies were provided by seven project cities: Basel (Switzerland), Kuopio (Finland), Rotterdam (Netherlands), Stuttgart (Germany), Suzhou (China), Thessaloniki (Greece) and Xi`an (China). Three policy areas affecting climate, energy and GHG emissions and largely influenced by local authorities were considered within the URGENCHE project framework: energy production and distribution (e.g. use of renewables), transport (e.g. modal split, traffic reduction, electric cars), and housing (e.g. energy efficiency, siting) (4,5).

Standard HIA approaches were applied within URGENCHE, quantifying impacts of particularly transport policies and related interventions on levels of air pollution, noise and physical activity and their ensuing implications for health (6,7). For the assessment of wellbeing impacts, no standard approaches exist and this paper presents exploratory results of URGENCHE work on wellbeing impacts of urban transport interventions in Basel (Switzerland), Rotterdam (The Netherlands) and Thessaloniki (Greece). Focus was put on noise rather than air pollution as noise perception is more likely to affect subjective wellbeing than air pollution which – unless pollution levels are very high - may be more difficult to be assessed by individual residents. The selection of noise also acknowledges that road traffic noise is considered the most widespread noise source in Europe, and a rising urban challenge associated with wellbeing (8).

Existing evidence on road traffic noise and wellbeing suggests a possible association, but the evidence on wellbeing effects is limited in comparison to health outcomes and tends to be based on local studies or specific context (see e.g. Lercher and Kofler, 1996 (9)). No validated and generally

applicable risk ratio or exposure-response function is available (see Supplementary file 1 for a short resume of literature). Given these limitations, we explored how to perform alternative wellbeing assessments of urban transport policies with available data on noise and wellbeing.

2. Data and methods

For assessing the impact of local interventions and the related traffic noise exposure changes on wellbeing, data on noise exposure before and after the intervention are needed. The year 2010 was selected as the starting point of the assessment (Baseline2010) and 2020 as the point in time when the effects of the intervention will be measured (Intervention2020). To account for other changes occurring between 2010 and 2020, a Business-as-Usual scenario (BAU2020) was established, including changes and developments from 2010 to 2020 that are independent of the planned intervention.

2.1 Data: City interventions

The transport-related interventions affecting noise exposure are described in Table 1 together with the respective Business-as-Usual scenarios).

Table 1. *Interventions to be implemented in case study cities by 2020*

Basel	*A scenario was developed by the city that accounts for additional local transport measures beyond a Business-as-Usual scenario (BAU2020) to further reduce private car road traffic by 4% on inner roads. It includes traffic measures targeted at channelling traffic along main avenues, reducing traffic levels and enforcing moderate speed limits in residential areas.* *The BAU2020 scenario in Basel does already include a range of measures adopted by the city by 2010 and implemented before 2020 (tram line extensions, expanding t capacity of main highways). An overall increase of 8% of vehicle kilometres has been estimated by 2020.*
Rotterdam	*50% of local car fleet will be electric cars (excluding motorbikes, vans and trucks).* *The BAU2020 scenario does not include specific interventions but accounts for expected changes in fleet composition related to Euro emission classes. It assumes zero growth of the traffic volume at inner-urban roads and 2% growth of the traffic volume at motorways as compared to the Baseline 2010 situation.*
Thessaloniki	*A local metro built in central Thessaloniki will reduce private road transport by an expected 33-44% in the city center and by 22% on main road axes leading to suburbs. The Intervention2020 scenario also includes a higher share of diesel and hybrid, but only a small (2%) share of electric vehicles in the fleet compared to the Baseline 2010.* *The BAU2020 scenario does not include specific interventions and serves as an extrapolation of the Baseline 2010 situation.*

2.2 Data for wellbeing impact assessments of urban interventions

Transport interventions for GHG reduction purposes are also expected to lead to reductions in noise exposure, which may have a positive effect on wellbeing. The change of overall wellbeing of the urban

population can be quantified by assessing wellbeing before and after the interventions, indicating to what extent the population will benefit from the urban GHG reduction measures in terms of wellbeing. For such an assessment, data are required for (a) noise exposure before and after the intervention, and (b) wellbeing impacts of noise exposure.

2.2.1. Data on road traffic noise exposure on city level

Change in noise exposure was calculated comparing the noise levels of the Intervention2020 and BAU2020 scenarios to the noise levels in the 2010 baseline scenarios. Noise level data were obtained from locally developed models for each city, using the respective standard methodology applied for noise mapping and modeling (see Supplementary file 2 for details). Data were provided in the format of Lden at façade or building entrance points. Lden is a weighted noise average over 24h, assigning higher weights to the evening and night periods than to the day period (following European noise regulation (10)). For all cities and scenarios, the fraction of population exposure by 1dB Lden was calculated.

2.2.2. Data on wellbeing impacts of noise

As no local data on noise and wellbeing associations were identified, national data were used as proxy instead. Noise and wellbeing data applied for Rotterdam and Thessaloniki were taken from the European Quality of Life Survey 2012 (EQLS2012) (11), and data applied for Basel were taken from the 2012 wave 14 of the Swiss Household Panel (SHP2012) (12). EQLS2012 data cover both perceived noise exposure (no, moderate and major noise problems) and the WHO_5 wellbeing index (13,14), enabling the quantification of associations between noise perception and wellbeing. The WHO_5 results in a score from 0 to 100 and we chose a cut-off of 52 to represent a binary outcome of low and high well-being as suggested elsewhere (13,15). The SHP2012 includes questions on noise annoyance with a dichotomous response (yes/no) and self-assessed mental wellbeing on a scale from 0 (never) to 10 (always); a score $\geq$6 was chosen as the cut-off to characterize individuals in low wellbeing. Both EQLS2012 and SHP2012 include relevant confounders of the noise-wellbeing association (such as age, gender, income, education), and allow rural residents to be identified and excluded from analysis. Table 2 provides an overview of the data components selected for the wellbeing assessment.

Table 2. Data used for the wellbeing impact assessment

City	**Urban noise exposure changes**	**Association between urban noise perception and wellbeing**		
		Data source	**Noise variable**	**Wellbeing variable**
Basel	Local noise models (Lden)	Swiss Household Panel 2011, urban residents (n=4505)	Annoyed by noise from neighbours or noise from the street (traffic, business, factories etc.). (Yes – No)	Do you often have negative feelings such as having the blues, being desperate, suffering from anxiety or depression? (Scale from 0 = "never" to 10 = "always")

Rotterdam		EQLS2012, Dutch urban residents (n=582)	Thinking of your immediate neighbor-hood - do you have problems with noise? (Major problems – Moderate problems – No problems)	WHO-5 wellbeing index (5 items producing a scale from 0 to 100)
Thessaloniki		EQLS2012, Greek urban residents (n=631)		

2.3 Methods for wellbeing impact assessment

Step 1 – Derivation of noise-wellbeing association using national proxy data

To avoid influence from rural noise conditions, each national dataset was reduced to urban-only residents. Differences between national and urban-only data were rather modest for age and gender as well as for noise and wellbeing levels, except for Greece where noise problems are more frequently reported in urban areas (see Supplementary Table 2).

Binary logistic regression models were ran, with wellbeing as response variable, and including the following independent variables: gender, age, income, education, employment, financial situation and household structure. Standardized residuals, group membership and predicted wellbeing probability were saved. For each regression model, the standardized residuals were checked for outliers which were deleted and the regression repeated.

The predicted wellbeing probability score covers a range from 0 (indicating the lowest probability of wellbeing) to 1, indicating the highest probability of wellbeing after taking into account the respective covariates (16). The wellbeing probability score can also be expressed in %.

Mean values of predicted wellbeing probability were calculated for the noise perception categories (no, moderate and major noise problems in Greece and the Netherlands and noise annoyance versus no noise annoyance in Switzerland) to define the wellbeing probability of the subpopulation in the respective noise perception category. These predicted probability values for the noise categories represent the “noise-wellbeing-relationship” that is later-on used for the wellbeing assessment of the URGENCHE interventions in the cities.

Step 2 - Transfer of EQLS and SHP noise perception to local noise models for case study cities

The wellbeing effect of the urban transport interventions is assessed by combining the modeled city data on noise exposure changes with the noise-related wellbeing probability (derived from the EQLS2012 and SHP2012 data). The combination is based on the assumption that EQLS2012 and SHP2012 data indicate the population percentage complaining about noise problems, and this percentage is applied to define the Lden ranges associated with certain noise perception categories in the respective city noise models (see details in Supplementary File 3). The connection of local noise exposure data with the noise perception data from EQLS2012 and SHP2012 results in cut-offs for low, medium and high noise perception tabled below which are slightly different in each city (Table 3).

Table 3. Noise perception category ranges for the city noise exposure profiles

Noise perception	Basel	Noise perception	Thessaloniki	Rotterdam
High: Annoyed by noise	>=64dB	**High: Major problems**	>=65dB Lden	>=67.5dB Lden
*	*	**Medium: Moderate problems**	55-64dB Lden	57.5-67.4dB Lden
Low: Not annoyed by noise	<64dB	**Low: No problems**	<=54 dB Lden	<57.5 dB Lden

**The SHP2011 dataset used for Basel does only distinguish between the perception of a noise as annoying or not annoying. It did not offer an intermediate option.*

Noise exposure changes in the different city scenarios lead to a changed percentage of the population predicted to report high, medium or low noise perception in the 2020 scenarios. As each noise perception category is associated with a specific probability of wellbeing, changes in population wellbeing can be finally estimated for each case study city and scenario.

Step 3 – Exploratory wellbeing impact assessments for the less affluent population

As research and political attention has increasingly focused on equity effects of public health and environmental interventions, wellbeing probability predictor values were also produced for the less affluent city population (defined as the two lowest income quartiles). Based on these values, we repeated the wellbeing impact assessment of noise reduction effects triggered by the urban interventions for the less affluent populations in each city.

3. Results

3.1 Noise exposure changes associated with the interventions

While the noise model predicts almost no noise variations between the scenarios in Rotterdam, there is a marked reduction of population exposure to noise by the Intervention2020 scenario in Thessaloniki. In Basel, the BAU2020 provides a strong reduction of high noise levels while the Intervention2020 scenario actually increases again the prevalence of high noise levels (see Supplementary Figures 1a-c for details). The modeled change of the population percentage categorized into different noise perception categories is shown in Table 4.

Table 4. Changes of perceived noise exposure in the case study cities

Level of perceived noise exposure in BASEL	**Population exposed in Baseline2010**	**Population exposed in BAU2020**	**Population exposed in Intervention2020**
	Total population	**Total population**	**Total population**
High: Annoyance by noise (≥64dB)	19.8%	10.8%	13.9%
Low: No annoyance by noise (<64dB)	80.2%	89.2%	86.1%
Level of perceived noise exposure in ROTTERDAM	**Population exposed in Baseline2010**	**Population exposed in BAU2020**	**Population exposed in Intervention2020**
	Total population	**Total population**	**Total population**
High: Major noise problem ((≥67.5dB)	1.6%	1.9%	1.7%
Medium: Moderate noise problem (57.5-67.4dB)	19.5%	20.4%	19.7%
Low: No noise problem (≤57.4dB)	78.9%	77.6%	78.6%
Level of perceived noise exposure in	**Population exposed**	**Population exposed**	**Population exposed**

THESSALONIKI	in Baseline2010	in BAU2020	in Intervention2020
	Total population	Total population	Total population
High: **Major noise problem (≥65dB)**	15.2%	15.4%	8.9%
Medium: **Moderate noise problem (55-64.9dB)**	40.6%	40.5%	36.4%
Low: **No noise problem (≤54.9dB)**	44.2%	44.1%	54.7%

3.2 Wellbeing probability predictors

The wellbeing probability predictor values are shown in Table 5 for each city and noise perception category. The wellbeing probability of the total population differs strongly between the cities, with the lowest probability found in Thessaloniki (63.1%) and the highest probability found in Basel (91.8%).

For all cities, the lowest wellbeing probability is found for the population group reporting high levels of noise. However, only in Rotterdam there is a straight dose-response relationship indicating that less noise is associated with a higher wellbeing probability, while in Thessaloniki the highest wellbeing probability is actually found for the persons reporting moderate noise problems (although the difference between low and medium noise perception is marginal). In Basel, the effect of high noise perception on wellbeing is less strong than in Rotterdam or Thessaloniki.

Table 5. Wellbeing probability in relation to noise perception

Level of perceived noise exposure	Predicted wellbeing probability (in %)
BASEL	**Total urban population sample (n=4505)**
High (≥64dB)	89.8%
Low (<64dB)	92.4%
Total population	91.8%
ROTTERDAM	**Total urban population sample (n=487)**
High (≥67.5dB)	73.1%
Medium (57.5-67.4dB)	78.1%
Low (≤57.4dB)	80.0%
Total population	79.5%
THESSALONIKI	**Total urban population sample (n=424)**
High (≥65dB)	55.9%
Medium (55-64.9dB)	64.7%
Low (≤54.9dB)	64.1%
Total population	63.1%

3.3 Wellbeing assessment at urban level

Combining the changes in noise levels with the respective wellbeing probability for each city and scenario provides indications of wellbeing changes for the total city population, and for the less affluent city population (Tables 6-8). As only a small proportion of the population moves from high to medium or low noise perception, and the wellbeing probability increase is limited as well, the overall wellbeing changes tend to be marginal on population level. More pronounced wellbeing impacts are therefore only found in residents reporting high noise perception at baseline.

For the total population of Basel, the wellbeing probability only increases by 0.3% from 91.8% at Baseline2010 to 92.1% under Intervention2020. Yet, the BAU2020 already achieved a wellbeing

probability of 92.1%, indicating that the Intervention2020 scenario provides no additional impact (Table 6). Compared to the total population, a stronger wellbeing impact is found for the residents reporting high noise levels (increase of 0.8%). However, for those residents reporting high perceived noise levels, the BAU2020 was even more effective (probability of wellbeing increase by 1.2%) but this effect is partially lost in the Intervention2020 scenario.

Table 6. Wellbeing probability in Basel by noise levels

Basel	**Intervention implemented by 2020**	**Predicted wellbeing probability (in %)**		
		Baseline2010	BAU2020	Intervention2020
Total population	Local transport scenario Z9, reduction of traffic by 4% *(NB: BAU2020 also includes various transport measures)*	**91.8%**	**92.1%**	**92.1%**
High noise perception		89.8%	91.0%	90.6%
Low noise perception		92.4%	92.4%	92.4%

The Rotterdam results show a 50% electric local car fleet has no wellbeing impact for the local population (Table 7). The negligible change of wellbeing probability is caused by the fact that the population reporting high noise levels did not change much between the scenarios. Yet, for the 1.6% of the population reporting high noise perception at baseline, there is at least some increase in wellbeing probability.

Table 7. Wellbeing probability in Rotterdam by noise levels

Rotterdam	**Intervention implemented by 2020**	**Predicted wellbeing probability (in %)**		
		Baseline2010	BAU2020	Intervention2020
Total population	50% of car fleet are electric cars	**79.5%**	**79.5%**	**79.5%**
High noise perception		73.1%	73.1%	73.8%
Medium noise perception		78.1%	78.1%	78.2%
Low noise perception		80.0%	80.0%	80.0%

In Thessaloniki, the wellbeing probability increases by 3.7% for residents reporting high noise levels (Table 8). Despite this relatively strong impact, the wellbeing increase remains limited for the total population because medium noise levels were associated with a slightly higher probability of wellbeing than low noise levels, leading to a decrease of overall wellbeing of the population in response to noise reduction for medium to low perception levels. This impact counteracts the increase of wellbeing probability found for residents reporting high noise exposure levels, and limits the overall wellbeing probability increase to 0.5% for the total population.

Table 8. Wellbeing probability in Thessaloniki by noise levels

Thessaloniki	**Intervention implemented by 2020**	**Predicted wellbeing probability (in %)**		
		Baseline2010	BAU2020	Intervention2020

Total population	Local metro built in central Thessaloniki	**63.1%**	**63.1%**	**63.6%**
High noise perception		55.9%	55.9%	59.6%
Medium noise perception		64.7%	64.7%	64.6%
Low noise perception		64.1%	64.1%	64.1%

3.4 Wellbeing assessment for less affluent population

The results of the wellbeing assessment for the less affluent population (two lowest income quartiles) are summarized below. Detailed result tables can be found in Supplementary File 4.

In all three cities, the less affluent population reports high noise perception more frequently, yet the noise reduction effect of the BAU2020 and Intervention2020 scenarios seems to be a bit smaller than for the total population. The wellbeing probability is lower for the less affluent population in all cities, with the largest difference in Thessaloniki (52.0% for less affluent population versus 63.1% for total population). Residents reporting high noise perception have a lower wellbeing probability, as already observed in the total population.

Wellbeing probability changes follow the same pattern as for the total population. In Basel, there is a small wellbeing benefit for the Basel BAU2020 scenario (increase by 0.5%) rather than the Intervention2020 scenario, while in Thessaloniki the Intervention2020 scenario increased wellbeing probability by 0.7%. In Rotterdam, the intervention has no impact on wellbeing at all.

Another similarity to the results for the total population is that the largest wellbeing increase is found for persons reporting high perceived noise at baseline; the increase of wellbeing probability found for Thessaloniki residents reporting high noise levels at baseline (4.0%) is the strongest wellbeing benefit identified in all three cities. However, compared to the total population, there is no indication of a generally stronger wellbeing benefit for the less affluent population in all three cities.

4. Discussion

This paper has presented an exploratory assessment of wellbeing impacts of urban transport policies and their noise effects in absence of standardized risk ratios and local studies linking urban noise conditions with the wellbeing of citizens. It proposed and tested a methodology that aims to make pragmatic use of the data available and derived exploratory results, triggering discussion on the potential impact of urban interventions on wellbeing and highlighting the main challenges of wellbeing assessments in terms of data and methodology.

4.1 Main findings

The main findings of this exploratory wellbeing assessment can be summarized as below:

1) The expected noise exposure changes resulting from the urban transport interventions are rather limited in all three cities.
2) Wellbeing probability is consistently reduced by perception of high noise levels, although this varies a lot in the different cities.

3) Across all three cities, the noise-related increase of wellbeing probability associated with the urban transport interventions is marginal. The strongest increase in wellbeing probability is found for Thessaloniki (0.5%).
4) Larger wellbeing benefits are consistently found for residents reporting high noise levels at baseline. This is valid in all cities (especially Thessaloniki) and for total and less affluent populations.
5) Less affluent population groups do not seem to derive a stronger wellbeing benefit from the transport interventions than the total population.

The very modest impact of the interventions on wellbeing seems to be conservative in light of some results that have been published before in relation to noise impacts on wellbeing (e.g. OR 1.50 for loss of wellbeing (9)), life satisfaction (e.g. OR 0.68 for life satisfaction for noise levels above 55dB (9), or mental wellbeing (e.g. OR 1.34 for dissatisfaction with street noise (17)), and rather matches studies reporting a marginal effect on wellbeing or life satisfaction (18,19).

There are various reasons to explain the limited wellbeing impacts of the transport interventions. One explanation is that urban transport interventions may provide a rather limited noise reduction effect (see also Stansfeld et al 2009 on a road traffic intervention study (20)), which restricts their potential wellbeing benefit. The reduction of road traffic or use of electric cars may not lead to significant noise reduction effects, especially as the rolling noise related to the moving car (considered the main source of road traffic noise for cars at a speed above 40km/h (8)) would not be reduced by electric cars.

Another reason for the marginal wellbeing effects is the rather weak association of noise with wellbeing. Relevant wellbeing benefits were only found for the rather small fraction of total population reporting high noise levels at baseline, and thus do not have a large impact on the total population figures. This is in agreement with other studies suggesting that noise-related wellbeing changes are more likely to be found when limiting the sample to residents strongly affected by noise problems. Yet, it is one of the encouraging findings of this exploratory work that the wellbeing effect of the transport interventions may be mostly realized in residents most affected by noise.

Lastly, the BAU2020 scenarios may also affect the impact of the Intervention2020 scenarios on probability of wellbeing. This is documented by the specific case of Basel, which has adopted sustainable urban planning principles many years ago. The Basel BAU2020 thus includes a set of measures reducing transport and related noise emissions (extended tram lines; increased road capacity etc.), which already increase the wellbeing probability for the total population and reduce the impact of additional interventions. The Basel Intervention2020 scenario aimed to reduce traffic on inner roads (where residential density is likely to be lower) and channel more traffic along main roads, where many residents are located and thus is likely to contribute to the modeled increase of high noise levels (13.9% in Intervention2020 versus 10.8% in BAU2020) which are associated with a lower wellbeing probability.

Residents reporting high levels of noise derive a higher wellbeing increase, while no increased wellbeing benefit is found in less affluent residents. This suggests that infrastructural interventions to reduce noise can help to mitigate noise exposure-related inequalities in wellbeing - due to their specific

impact on persons severely suffering from noise -, but that such interventions may not be suitable to reduce social inequalities in noise exposure and address specific target groups.

4.2 Limitations

Although there are some methodological strengths (use of validated survey data assuring comparable data; enabling coverage of various relevant control variables as covariates, and providing a validated wellbeing tool in case of EQLS2012 data), a variety of methodological limitations and assumptions affect our study and the findings, making further research on wellbeing effects of urban interventions necessary.

The first and most important limitation refers to the lack of a consistent and undisputed wellbeing measurement tool and the choices made by our exploratory study to operationalize wellbeing. Using the WHO_5 was a pragmatic choice as it (a) was available in the EQLS2012 dataset for many European countries, (b) represents a validated tool based on five items rather than one question only, and (c) already comes with a recommended cut-off to distinguish between low and high wellbeing. For the Basel case study, the variable considered closest to the WHO_5 wellbeing tool focused more on mental wellbeing and is thus not fully comparable. In addition there was no default cut-off level to define high versus low wellbeing on the applied response scale from 0-10. An alternative approach to wellbeing measurement is the concept of "life satisfaction", which has been recommended by the WHO Regional Office for Europe and OECD as a measure of subjective wellbeing (21,22).

Second, there is little evidence on the link between noise and wellbeing. In the absence of validated risk estimates and due to lack of data on noise-wellbeing linkages on city level, national EQLS2012 and SHP2012 datasets were used as proxy instead. The wellbeing predictor values applied to the city-specific noise models therefore represent the average national noise-wellbeing association in urban residents in the respective country, rather than the association for the given city.

Third, the wellbeing impact assessment presented has applied noise perception data from EQLS2012 and SHP2012 to attach a predicted wellbeing probability value to different noise categories. The link between noise perception levels and wellbeing as derived from EQLS2012 and SHP2012 data is assumed to be applicable for all three scenarios. Further work would be needed to assess whether the presented approach may lead to over- or underestimation of the wellbeing impact.

Fourth, we have used cross-sectional datasets and thus it is impossible to indicate whether there indeed is a causal relationship between noise and wellbeing, or whether different levels of wellbeing possibly affect noise perception. This may especially be the case in Basel, where SHP2012 data on mental wellbeing was applied which may have affected the rating of noise exposure.

Fifth, there are limitations with the noise data applied for the wellbeing assessments. A general limitation is the use of Lden data which represents the overall noise level during day, evening and night but does not identify peak exposure levels. Although Lden is often used in studies on the long-term impacts of noise exposure and is also suggested as an indicator for noise maps required by the EU Environmental Noise Directive (10), it is unclear whether it is the most suitable noise indicator for wellbeing impacts. More specific to our study, another limitation is that each city provided their own

noise model and methodological differences introduced by local modeling approaches were unavoidable.

Sixth, the findings presented only indicate the wellbeing effects of noise exposure changes attributable to the interventions. The results do not include potential wellbeing effects triggered through other pathways (such as e.g. air pollution reduction); such effects are likely to exist and they may exceed the wellbeing effects in relation to noise (23). Thus, the results presented reflect by no means the total wellbeing impact of the described urban transport policies.

Acknowledging the relevance of these limitations, the results provided in this paper do not represent reliable findings. Instead, the merit of this paper may rather be in providing a first exploratory wellbeing impact assessment based on available data, and identifying the gaps of evidence for implementing a more thorough and valid assessment. The quality of the findings reflect the quality of data used and progress on the following aspects is needed to enable more reliable assessments of wellbeing impacts of urban interventions through:

- Valid, internationally comparable and consensus-based definition and compilation of wellbeing data;
- Derivation of reliable risk ratios for environmental conditions and wellbeing; and
- Local or national surveys and modeling approaches providing adequate baseline data and enabling the generation of models and future scenarios.

A first step towards establishing a recommended approach towards the measurement of subjective wellbeing has been taken by WHO Regional Office for Europe with the recent adoption of "life satisfaction" as a core indicator on subjective wellbeing for monitoring the Health2020 policy of the WHO European Region, aiming at the compilation of consistent wellbeing data from all European member states (22). On the impacts of noise on wellbeing, the forthcoming WHO Environmental Noise Guidelines for the European Region will summarize the existing evidence.

Further research that would move this scientific challenge ahead and enable HIA to also include notions of wellbeing would especially be longitudinal studies (e.g. taking measures of wellbeing in a city population before and after an intervention and record changes, also taking into account other factors), or field experiments (e.g. follow up on wellbeing of e.g. randomly selected residents living in areas highly exposed to traffic versus quiet areas and measure wellbeing over a longer time period).

4. Conclusions

The findings from the three case study cities suggest that the noise reduction effects of urban transport interventions are limited but may still trigger slight wellbeing benefits. The fact that urban transport interventions may have positive impacts on wellbeing should not be taken for granted, especially as wellbeing was not explicitly considered a relevant factor for selection and implementation of the city interventions (which actually focused on climate change mitigation). However, the results also indicate that within a given city population, wellbeing benefits are mostly realized within residents reporting high noise levels, while there is no additional wellbeing benefit for less affluent residents when compared with the total population. This is an important indication in

relation to potential equity effects of urban interventions, but needs to be substantiated by further studies.

This paper showed that despite various methodological constraints and gaps of evidence, an exploratory assessment of wellbeing effects of urban climate change mitigation policies is possible even though the results are relatively indicative. However, the merit of this work is rather in identifying the missing components for a more accurate and valid wellbeing assessment and provoking further research in response to this paper, making wellbeing assessments increasingly reliable. Especially relevant would be the derivation of exposure-response functions for specific environmental determinants and wellbeing.

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Author Contributions

MB conceived the assessment approach together with MT, and drafted the first version of the paper. PM, MM, RH and DC contributed to further development of the approach. RH and MB compiled and reviewed relevant literature. DS and MK provided the noise model data for Thessaloniki and Rotterdam, respectively. MB, MT, DC and PM provided the wellbeing assessment for Thessaloniki and Rotterdam. LP conducted the wellbeing assessment for Basel. MB, PM, MT, LP and RH contributed to the discussion of approach and findings and all authors contributed to the final version of the paper.

Conflicts of Interest

The authors declare no conflict of interest.

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Supplementary file 1: Summary of studies on road traffic noise and wellbeing

The reviewed evidence on road traffic noise and wellbeing did not allow deriving valid and reliable "ratios" of wellbeing status associated with certain levels of noise exposure. Only four studies were identified which presented associated measures of road traffic noise with a measure of subjective wellbeing (see Supplementary Table 1). Only two of the four studies involved a larger sample size, one study focused on a very specific setting (rural alpine communities affected by through-traffic noise), and one study was restricted to one city only. All studies were cross-sectional and only two included control variables in the analysis. Finally, two out of four studies did not provide significant results on wellbeing impacts of road traffic noise.

Supplementary Table 1: Papers presenting quantitative associations between noise and wellbeing measures

Author	Noise and wellbeing measures	Setting	Study design / sample size	Con-trols	Association	Sig	Restriction
Lercher/Kofler, 1996 (9)	Noise level from traffic Loss of wellbeing; Life satisfaction	Five rural alpine communities, Austria	Cross-sectional; n=1989	Age, Sex, SES	Loss of wellbeing (dichotomized) OR 1.50 (1.14-1.96) above 55 decibels vs 55 or lower; Life satisfaction (dichotomized) OR 0.68 (0.51-0.90) above 55 decibels vs 55 or lower	Yes	Specific setting unlikely to reflect urban noise conditions
Schreckenberg et al., 2010 (18)	Daytime noise level (road) Life satisfaction (FLZ score)	Frankfurt Germany	Cross-sectional, n=190	None	Life satisfaction coefficient of correlation= .103	No	Small sample, one city only
Urban/Maca, 2013 (19)	Road noise (strategic noise maps) Life satisfaction	5 Czech cities	Cross-sectional; n=354	None	Life satisfaction r=.066	No	Small sample, Czech data only
Rehdanz/Maddison, 2008 (23)	Perceived local noise nuisance Life satisfaction	Germany	Cross sectional; n=23000	Unclear	For each step reduction in feeling adversely affected by noise (5 steps), a person is .85% more likely to score highest life satisfaction levels, .63% less likely to score average life satisfaction levels, and .34% less likely to score lowest life satisfaction levels.	Yes	German environmental preference data only

Other noise-specific studies providing quantitative risk ratios were identified, but these focused on more health-specific outcomes such as mental health (17,18,20,24–26), annoyance (8,9,18,19,27–31), or health-related quality of life (32,33). As well, some studies referred to wellbeing-related impacts of e.g. aircraft noise or railways (18,19,30,34–36) but these were not considered applicable for road traffic. Hiscock et al. (2014) have indicated in this context that risk ratios for noise and wellbeing are rare mainly because of the following points: insufficient quality and quantity of available studies; diversity of measures on wellbeing outcomes; and the rather varying risk ratios indicated by the studies

(for strongly different population groups in rather different local contexts) (5). During our literature review, we also noted that there is mixed evidence regarding the equity dimension of noise exposure, with some studies pointing to increased noise exposure in disadvantaged population groups (37–39) and some studies suggesting the opposite (40,41), but no study looking at wellbeing impacts of noise exposure from an equity lens.

Supplementary file 2: Local noise data models and restrictions of comparability

The noise data used for the Basel and Rotterdam scenarios was provided by the databases used by the respective local authorities for decision-making in urban planning, while for Thessaloniki the noise modeling has been produced for the URGENCHE project and the model results have been validated against field measurements in two stations (years 2004, 2011 and 2012). Modeling methods differed between the cities as summarized below, but it is not possible to assess to what extent the different approaches may have affected the results.

Basel: The noise exposure model for Basel was provided by the municipality of Basel and takes into account all road traffic (individual motorized traffic as well as tram and bus lines and their frequencies). Noise levels at several façade points were developed using the emission obtained from the local road traffic models and a noise propagation model (CADNA) that link source of emissions to reception points. The model considered building height, first order reflections from building facades, and noise barriers such as e.g. public greenery (7).

Rotterdam: The noise exposure model for Rotterdam only accounts for road traffic noise and excludes other urban noise sources (train, aircraft, trade and industry). The road traffic noise exposure of the subjects was calculated at the most- and the least exposed facade of the given dwelling with the Dutch standard method SRM2 in accordance with requirements of the EU Environmental Noise Directive (END). The noise calculations are based on road traffic characteristics, including traffic intensity, traffic composition (percentage of light duty, medium duty, and heavy duty vehicles) and speed, and take into account the effects of buildings on propagation of noise.

Thessaloniki: The noise model provided by the Aristotle University of Thessaloniki accounts for road traffic noise covering any road transport. The noise calculations are based on road traffic characteristics (such as traffic intensity) as well as road infrastructure features. Given that the noise impact calculations are clustered at the municipality level, variable residual uncertainty remains around the estimated value on the basis of the urban landscape in each municipality in the Thessaloniki metropolitan area. These uncertainties would tend to slightly underestimate the noise level in 2020.

Supplementary Table 2: Noise and wellbeing in the case study cities and countries

Noise and wellbeing in Switzerland (Source: SHP2012)		
Noise perception	**Switzerland, total**	**Switzerland, urban**
Annoyed	20.7%	22.4%
Not annoyed	79.3%	77.6%
Total n	6014	4505
Wellbeing	**Switzerland, total**	**Switzerland, urban**
High wellbeing	92.0%	91.8%
Low wellbeing	8.0%	8.2%
Total n	6014	4505
Noise and wellbeing in the Netherlands (Source: EQLS2012)		
Noise perception	**Netherlands, total**	**Netherlands, urban**
Major problems	1.8%	1.7%
Moderate problems	18.1%	21.0%
No problems	80.2%	77.3%
Total n	1008	582
Wellbeing	**Netherlands, total**	**Netherlands, urban**
High wellbeing	79.4%	77.0%
Low wellbeing	20.6%	23.0%
Total n	1005	582
Noise and wellbeing in Greece (Source: EQLS2012)		
Noise perception	**Greece, total**	**Greece, urban**
Major problems	9.6%	14.4%
Moderate problems	31.6%	40.2%
No problems	58.7%	45.4%
Total n	1003	630
Wellbeing	**Greece, total**	**Greece, urban**
High wellbeing	65.7%	64.5%
Low wellbeing	34.3%	35.5%
Total n	1002	629

Supplementary File 3: Establishing noise cut-offs on city level on the basis of national datasets

The matching of the local noise model data and the noise perception categories derived from EQLS2012 and SHP2012 data is explained below using the example of Thessaloniki.

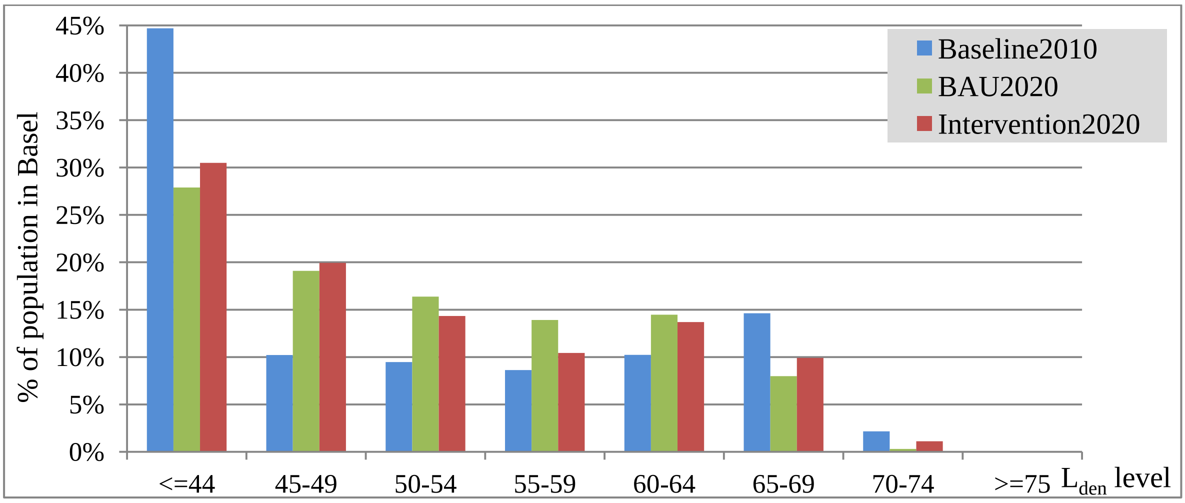
The EQLS2012 indicated that in Greece, about 14.4% of the urban population report major problems with noise. Transferring these 14.4% to the modeled noise exposure data from Thessaloniki would suggest 65dB Lden as the most suitable noise cut-off of, as 15.2% of Thessaloniki`s population are exposed to 65dB Lden and beyond. This is the noise level affecting the population percentage closest to 14.4% (cut-off levels at 64 and 66dB would be less close to the 14.4%). The same approach was applied for moderate and no problems with noise.

Supplementary Table 3. Derivation of noise cut-offs for the city noise exposure profiles - Thessaloniki example

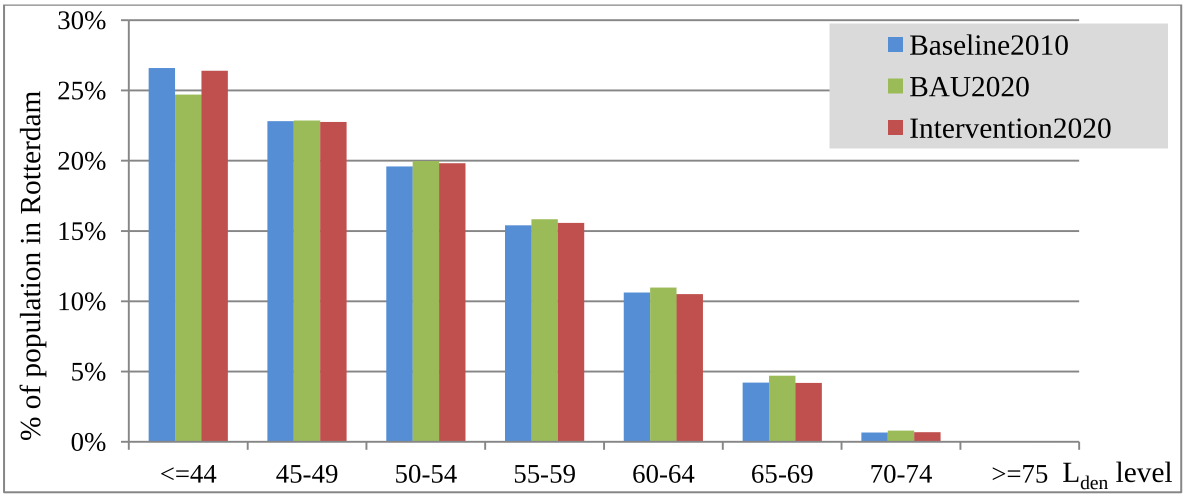
Noise perception	EQLS2012 data, urban Greece	Closest matching exposure range for Thessaloniki noise model	
Major problems	14.4%	15.2%	>=65dB Lden
Moderate problems	40.2%	40.6%	55-64dB Lden
No problems	45.4%	44.2%	<=54 dB Lden
Total n	630 persons	344 244 persons	

Supplementary Figures 1a-c

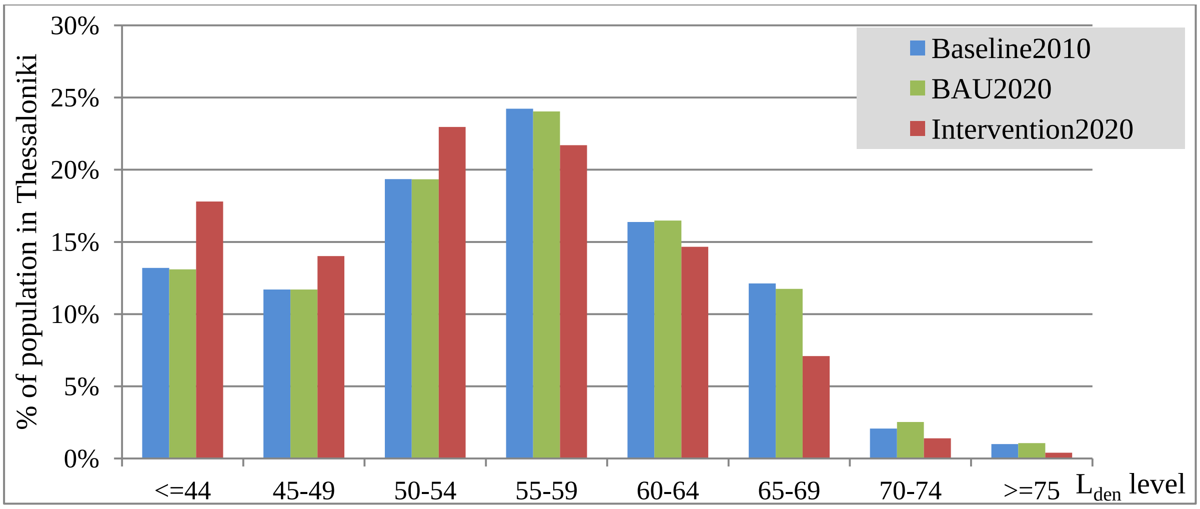
Supplementary Figure 1a: Comparison of noise exposure distribution in the population of Basel at Baseline2010 and under BAU2020 and Intervention2020



Supplementary Figure 1b: Comparison of noise exposure distribution in the population of Rotterdam at Baseline2010 and under BAU2020 and Intervention2020



Supplementary Figure 1c. Comparison of noise exposure distribution in the population of Thessaloniki at Baseline2010 and under BAU2020 and Intervention2020



Supplementary file 4: Detailed result tables on wellbeing changes in the less affluent population

Supplementary Table 4: Changes of perceived noise exposure in the less affluent city population

Level of perceived noise exposure in BASEL	Population exposed in Baseline2010	Population exposed in BAU2020	Population exposed in Intervention2020
	Less affluent population	Less affluent population	Less affluent population
High: Annoyance by noise (≥64dB)	24.1%	16.9%	19.0%
Low: No annoyance by noise (<64dB)	75.9%	83.1%	81.0%
Level of perceived noise exposure in ROTTERDAM	**Population exposed in Baseline2010**	**Population exposed in BAU2020**	**Population exposed in Intervention2020**
	Less affluent population	**Less affluent population**	**Less affluent population**
High: Major noise problem ((≥67.5dB)	2.5%	2.9%	2.5%
Medium: Moderate noise problem (57.5-67.4dB)	18.6%	19.5%	18.9%
Low: No noise problem (≤57.4dB)	78.9%	77.6%	78.6%
Level of perceived noise exposure in THESSALONIKI	**Population exposed in Baseline2010**	**Population exposed in BAU2020**	**Population exposed in Intervention2020**
	Less affluent population	**Less affluent population**	**Less affluent population**
High: Major noise problem (≥65dB)	19.9%	20.2%	15.2%
Medium: Moderate noise problem (55-64.9dB)	39.0%	38.9%	41.4%
Low: No noise problem (≤54.9dB)	41.1%	40.9%	43.4%

Supplementary Table 5: Wellbeing probability in relation to noise perception for less affluent population groups

Level of perceived noise exposure	Predicted wellbeing probability (in %)
BASEL	**Less affluent urban population sample (n=2249)**
High (≥64dB)	86.7%
Low (<64dB)	90.6%
Total population	89.5%
ROTTERDAM	**Less affluent urban population sample (n=245)**
High (≥67.5dB)	67.8%
Medium (57.5-67.4dB)	70.2%
Low (≤57.4dB)	74.9%
Total population	73.8%
THESSALONIKI	**Less affluent urban population sample (n=198)**
High (≥65dB)	40.6%
Medium (55-64.9dB)	56.5%
Low (≤54.9dB)	53.3%
Total population	52.0%

Supplementary Table 6: Wellbeing probability by noise levels - less affluent population

Basel	**Intervention implemented by 2020**	**Predicted wellbeing probability (in %)**		
		Baseline2010	BAU2020	Intervention2020
Less affluent population	Local transport scenario Z9, reduction of traffic by 4% *(NB: BAU2020 also includes various transport measures)*	**89.5%**	**90.0%**	**89.9%**
High noise perception		86.7%	87.9%	87.5%
Low noise perception		90.6%	90.6%	90.6%
Rotterdam	**Intervention implemented by 2020**	**Predicted wellbeing probability (in %)**		
		Baseline2010	BAU2020	Intervention2020
Less affluent population	50% of car fleet are electric cars	**73.8%**	**73.8%**	**73.8%**
High noise perception		67.8%	67.8%	68.1%
Medium noise perception		70.2%	70.2%	70.4%
Low noise perception		74.9%	74.9%	74.9%
Thessaloniki	**Intervention implemented by 2020**	**Predicted wellbeing probability (in %)**		
		Baseline2010	BAU2020	Intervention2020
Less affluent population	Local metro built in central Thessaloniki	**52.0%**	**52.0%**	**52.7%**
High noise perception		40.6%	40.6%	44.6%
Medium noise perception		56.5%	56.5%	56.3%
Low noise perception		53.3%	53.3%	53.3%

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The authors declare that they have no actual or potential competing financial interests.

Dear Editor,

Please find attached our manuscript that assesses the health impacts of two greenhouse gas mitigation policies in Rotterdam, Netherlands. The research presented was derived from an international research collaboration conducted within the context of the project “Urban Reduction of GHG Emissions in China and Europe (URGENCHE)”.

We believe our method and our findings could be of interest for your readers, because policy relevant measures were assessed of their co-benefits on health. We would greatly appreciate the opportunity for this manuscript to go into peer review.

Hereby we assure that the present manuscript is an original work, which was not previously published and is not under consideration for publication elsewhere. Merely preliminary findings were presented at the 26th Annual International Society for Environmental Epidemiology Conference 2014**.** No conflict of interest is reported from any author. All authors have read the manuscript, agree the work is ready for submission, and accept responsibility for the manuscript contents.

We thank you for your consideration of our manuscript.

Sincerely,

M. Tobollik

Myriam Tobollik

Abstract

Background: Greenhouse gas (GHG) mitigation policies are increasingly considered to minimize negative effects of climate change. Health impact assessment (HIA) of transport policies is a useful tool to evaluate their consequences on noise and air quality and co-benefits on health.

Objective: The Rotterdam case study of GHG mitigation policies regarding road traffic provided data to assess their health effects.

Methods: HIA, modeling population exposure to air pollution and noise and quantifying the expected burden in Disability-Adjusted Life-Years (DALY), was conducted for two planned traffic policies to be effective by 2020: a) 10% traffic reduction; b) 50% electric car use.

Results: A comparison of the baseline situation to implementation of both policies results in 2,097 (UI (Uncertainty Interval) 1,403, 2,711) life years lost for total mortality, 346 (UI: 269, 413) life years lost for ischemic heart disease mortality and 338 (UI: 138, 504) life years lost for lung cancer mortality which can be prevented by a decrease of annual average $PM_{2.5}$ from 19.95 $\mu g/m^3$ in 2010 to 9.51 $\mu g/m^3$ in 2020. Noise exposure is supposed to increase, still comparing the policies with a business as usual scenario would show positive effects.

Conclusions: The case of European cities that have already high environmental standards is of interest, while the results can be rather small, they can be used to discuss public health issues related to planning and to inform on the co-benefits of GHG mitigation policies. Innovative HIA estimates should include Elemental Carbon to consider road traffic intensity.

Introduction

Recently there have been many efforts to reduce greenhouse gas (GHG) emissions at national and international level (for example Kyoto Protocol, Directive 2003/87/EC, Directive 2009/29/EC, Lima Climate Change Conference). Beside energy production and the building sector, the transport sector is a leading contributor to carbon dioxide (CO_2) emissions. It represented around 22% of worldwide CO_2 emissions in 2010 (International Energy Agency 2012).

Decisions concerning measures in the transport sector to reduce GHG emissions also affect air quality and noise (Haines et al. 2009). Likewise, they can affect health in an intended or unintended manner, as well as positive or negative directions (Kemm and Parry 2004; Mindell et al. 2003). Thus, there is the need to integrate in traffic modeling an Health Impact Assessment (HIA) of air pollution and noise exposure and related health effects (WHO European Centre for Health Policy 1999).

Several recent studies have addressed the impacts of transport policies on health (e.g. Dhondt et al. 2013; Hosking et al. 2011; Thomson et al. 2008). Transport can be responsible for several adverse health effects, such as increased risk of cardiovascular or respiratory disease due to air pollution (e. g.Brunekreef and Holgate 2002; Hoek et al. 2013; Künzli et al. 2000; Pope and Dockery 2006), injuries caused by dangerous driving behaviour (Peden et al. 2004) or annoyance caused by traffic noise (Miedema et al. 2011; Miedema and Oudshoorn 2001).

The city of Rotterdam, located 20 km from the coast of the North Sea in the west of The Netherlands, has planned a series of measures to decrease CO_2 emissions by 50% between 1990 and 2020 as part of its GHG mitigation policies. These measures include more biomass burning in energy production, insulation of buildings to reduce the demand for energy and traffic related

measures. The latter relates to a) 10% fewer private vehicle kilometers on inner-urban roads and b) 50% electric car use in year 2020. These measures support the attempt of the city to become clean, green and economically robust by reducing noise levels and "improving air quality, particularly in view of the influence this has on human health" (Rotterdam Office for Sustainability and Climate Changes 2011:6).

The goal of our study is to present an analysis of health co-benefits of two GHG mitigation policies in the transport sector in Rotterdam. Effects of these measures were evaluated for year 2010, chosen as a baseline and the year 2020 was selected as the end of the assessment where the impact of the measures is compared to a business-as-usual (BAU) development. The BAU scenario includes the assumptions that the consequences of today's exposure and behavior will continue without any changes into the year 2020, furthermore it includes regulations which are already decided but not yet implemented, like the exhaust emission standard Euro 6.

Methods

Health Impact Assessment

Exposure measurements for air pollution and noise related to the transport scenarios were produced. These factors in turn were used to calculate effects on health and well-being of the population in Rotterdam. Air pollution has been assessed by its impact on all-cause mortality and more specifically on ischemic heart disease and lung cancer mortality and morbidity (for example restricted activity days, RAD), while the latter has been assessed by its impact on serious annoyance and sleep disturbance.

The health impact of the policies has been expressed in Disability-Adjusted Life Years (DALY), because to "facilitate comparison, health effects are best expressed not only as the number of people affected, but also in summary measures of population health" (Schram-Bijkerk et al.

2009: 691). This summary measure of population health includes both mortality (Years of life lost, YLL) and morbidity (Years lived with disability, YLD) in one measurement (Knol et al. 2009; Murray et al. 2002).

The health impacts of the transport measures were estimated through four steps: 1) identifying transport-related effects in terms of risk factors for health; 2) modeling the spatial distribution of noise and air quality, and related population exposure according to the transport scenarios; 3) selecting health outcomes related to the risk factors and relative risks; 4) quantifying the expected health burden, expressed in DALYs, which can be prevented by introducing the two policies.

Step 1: Identification of the impacts of the traffic policies

For air quality, particulate matter $(PM)_{2.5}$ has been identified as an adequate indicator for all-cause mortality due to long-term exposure (WHO 2006). It should be noted that there is no threshold level below which no negative effects are expected. The World Health Organization (WHO) has indicated in the air quality guidelines that the average annual level of $PM_{2.5}$ should not be higher than 10 $\mu g/m^3$ to protect public health.

Primary emissions of soot particles from road traffic contribute very little to the mass of $PM_{2.5}$ and consequently, $PM_{2.5}$ is not a sensitive indicator for local traffic emissions (Keuken et al. 2012). In view of the notion that there is a need to characterize “health risks of air pollution near sources of combustion particles” (NA Janssen et al. 2011: 1691), black or elemental carbon has been suggested as additional indicator. This is also recognized by the WHO: “Studies taking black carbon and $PM_{2.5}$ into account simultaneously associations remained robust for black carbon. Even when black carbon may not be the causal agent, black carbon particles are a valuable additional air quality metric to evaluate the health risks of primary combustion particles

from traffic" (WHO European Centre for Environment and Health 2013: 10). In the HIA for Rotterdam, both $PM_{2.5}$ and Elemental Carbon (EC) have been examined to assess the impact of the two transport measures on health.

In addition to air pollution, road traffic also causes noise. Noise levels increase with higher traffic volumes and speed (Hosking et al. 2011). For traffic noise, a weighted average over 24 hours is performed, assigning higher weights to the evening and night periods than to the day period. The weighted average noise levels are called "L_{den}" (day-evening-night) and "L_{night}" (only during the night). L_{den} is associated with annoyance and hence an indicator for well-being, while L_{night} is related to sleep disturbance and cardiovascular effects (Miedema et al. 2011). In the HIA for Rotterdam high noise annoyance (people exposed to L_{den} over 55 dB(A)) and sleep disturbance due to noise (people exposed to L_{night} over 50 dB(A)) has been used to assess impacts of the two traffic measures.

Step 2: Modeling the exposure to transport measures

Modeling the spatial distribution of air quality ($PM_{2.5}$, EC) and noise (L_{den}, L_{night}) requires various input data: the road and motorway network in Rotterdam, the related traffic (i.e. volume, fleet composition and speed), meteorology (wind speed and direction), background concentrations of air quality and buildings near roads and motorways. The road network and traffic data are available from the traffic department in the city of Rotterdam. The total road traffic volume in 2010 was 13 million vehicle kilometers per 24 hours with 68% and 32%, respectively on inner-urban roads and urban motorways. The average distribution of personal cars, light duty and heavy duty trucks is, respectively 90%, 5% and 5%. The expected volume growth of road traffic by Rotterdam authorities till 2020 is zero on inner-urban roads and 2% per year on urban motorways. For dispersion of air pollutants at inner-urban roads in Rotterdam, a

"street canyon" model was applied and for motorways a "line-source" model (Beelen et al. 2010; Keuken et al. 2012). In addition to the aforementioned data input, the dispersion models also require emission factors for road traffic in 2010 and 2020 for primary $PM_{2.5}$ and EC. Secondary $PM_{2.5}$ is also included by NOx and NO_2 formation which is located outside of the city and thus is part of the background concentration of $PM_{2.5.}$ The different components of $PM_{2.5}$ were not considered in this assessment. However, they can have different effects on health (Hurley et al. 2005). Thus more information is needed to assess these effects as well as their combined effects to avoid double counting. So far a single-pollutant model is preferred, because the effects of $PM_{2.5}$ seems to be robust to adjustment for other pollutants (WHO European Centre for Environment and Health 2013).

Data are available on a national level and are based on the European emissions inventory for road traffic "COPERT" (Ntziachristos and Samaras 2009; Velders et al. 2012). The contribution of road traffic emissions is added to the background air quality concentrations and presented on a Geographical Information System-map with the road network of the city. The background concentrations of primary and secondary $PM_{2.5}$ and EC in 2010 and 2020 at a spatial resolution of 1*1 km are available in the Netherlands (Velders et al. 2012).

Noise calculations in Rotterdam have been carried out in two steps: firstly, calculating the emission and then the transmission. The emission calculations take into account traffic and road characteristics: traffic intensity, traffic composition (percentages light duty, medium duty, and heavy duty vehicles), speed, road height, and road surface. Similar to air quality, emission factors have been established in the Netherlands for road traffic and distinguishing emissions from tyre and engine noise. The transmission calculations take into account the distance between source (road) and building facade, air and ground attenuation, annual average meteorological

conditions and reflection of objects opposite the building. These calculations result in the noise exposure at the center of a building represented in decibels (dB). The noise exposure in Rotterdam was calculated with the Dutch standard method (SRM2). SRM2 is in accordance with requirements of the EU Environmental Noise Directive (European Parliament and Council of the European Union 2002). Therefore, for $PM_{2.5}$, EC and L_{den} the exposure at residential addresses of the population was distinguished in people living in street canyons with more than 10,000 vehicles per 24-hours, people living within 100 m from a motorway and the rest of the population.

Step 3: Selection of health outcomes related to risk factors

Health outcomes of which there is evidence of a causal relationship with air pollution and road traffic noise were selected based on a literature search, which was focused on meta-analysis and recommendations of international organizations such as the European Commission or WHO (see Table 1). The health outcomes were selected due to availability of data on mortality and strength of available evidence. There is evidence that $PM_{2.5}$ also triggers morbidity outcomes, but it is not enough to quantify these effects. Especially for EC data are still limited, because the scientific interest has developed only in the last decades.

Table 1

The exposure response functions (ERF) were selected due to their accuracy (meta-analysis were preferred) and applicability (international ERFs were used because they are more robust than local ones). Nevertheless, by using international ERFs we assume that they are valid for the population of Rotterdam.

Besides the ERFs, demographic, mortality and prevalence data is needed. Demographic data on the population of Rotterdam stratified by sex and age (five-year age groups) for the year 2010

were obtained from the Gemeente Rotterdam Centrum voor Onderzoek en Statistiek webpage (2014) as well as all-cause mortality data on the same level of detail. Mortality and prevalence data were not available on local level. Therefore national prevalence rates were used by assuming that these rates are similar to local rates. The data were available online from Statline (2014) and IKNL (2014). All-cause mortality related to long-term exposure to $PM_{2.5}$ and EC was calculated for people aged 30 years and older. Furthermore lung cancer and ischemic heart disease related mortality caused by $PM_{2.5}$ were estimated. A causal relationship is also available for noise and the health outcomes annoyance and sleep disturbance.

To facilitate a comprehensive assessment not only health outcomes are included. In addition, restriction in activity because of ill-health was gathered in terms of Restricted Activity Days (RAD). A RAD is a day when a person is forced to alter his or her normal activity for health related reasons. These days are highly related to $PM_{2.5}$ exposure (Hurley et al. 2005; Ostro 1987).

Step 4: Quantification of the expected health burden

We used the population data of 2010 for the burden of disease calculation in the year 2020 to ensure a comparison of the 2010 and the future scenarios. Population estimates for Rotterdam predict a small increase in the population up to 640,215 in 2020 (Hoppesteyn 2012). The expected change for the number of people living in Rotterdam in the next 15 years is negligible.

For long-term projection discounting of time is suggested (Remais et al. 2014), but in our assessment the prevented life years in the future have the same weight as the ones today. Likewise we didn’t apply any age-weighting. For each health outcome the burden of disease attributable to the exposure was calculated stratified by 5 years age groups and sex. Excel was used to do the calculations (downloadable from the WHO website: http://www.who.int/healthinfo/global_burden_disease/tools_national/en/ [04.05.2014]).

First, the counterfactual concentration is calculated, which gives the difference between the measured air pollution concentration in the baseline scenario and the modeled concentration in the future scenario. In the present assessment the counterfactual concentration for $PM_{2.5}$ was 7.41 $\mu g/m^3$, calculated by subtracting 9.51 $\mu g/m^3$ $PM_{2.5}$ (concentration of the BAU and the two intervention scenarios) from 16.92 $\mu g/m^3$ $PM_{2.5}$ (concentration of the baseline scenario) and for EC the counterfactual concentration was 0.7 $\mu g/m^3$ (1.2 minus 0.5 $\mu g/m^3$).

Based on the ERF the proportion of population risk which is attributable to a risk factor was calculated (Prüss-Üstün et al. 2003). In the last step the death cases of the disease due to the exposure are multiplied by the life expectancy to obtain YLLs. The ERFs used were available together with confidence intervals, which were used to calculate the upper and lower confidence limits as part of a sensitivity analysis.

For noise the formulas presented in Step 3 were used to calculate the number of people annoyed and sleep disturbed. These numbers were then multiplied by a Disability Weight (DW) to obtain YLDs. The DWs from the Environmental Burden of Disease study on noise of the WHO and the recommendations of the European Commission were used: for annoyance 0.02 (lower bound 0.01 and upper bound 0.12 (Miedema et al. 2011)) and for sleep disturbance 0.7 (lower bound 0.04 and upper bound 0.1 (S Janssen et al. 2011)).

The possibly synergistic effects due to interactions of $PM_{2.5}$ and noise were not considered in the present HIA despite some preliminary indications that such interactions exists (Schram-Bijkerk et al. 2009). Given that no studies have quantified these potential interactions we could not use them in this assessment (Babisch et al. 2014) .

Results

As expected, people living near intense road traffic are more affected by transport policies than the rest of the population. The annual average concentrations for $PM_{2.5}$, EC, L_{den} and L_{night} in 2010 are shown in Figure 1 and in Table 2 and 5.

Figure 1

Figure 1a illustrates that $PM_{2.5}$ levels in the Netherlands and the rest of Europe (EEA 2012) are dominated by the regional background and there is limited contribution of emissions from road traffic near motorways around Rotterdam and along inner-urban roads in the center of Rotterdam. Figure 1b illustrates that EC is a more sensitive indicator for road traffic emissions than $PM_{2.5}$, as EC concentrations near intense road traffic are a factor of 2 to 3 higher compared to the urban background in Rotterdam. Figures 1c and 1d illustrate that noise emissions from road traffic result in large parts of the population being exposed to L_{den} levels over 55 dB in Rotterdam.

Table 2

The BAU 2020 will result in significantly lower air pollution as compared to 2010 (see Table 2). This is mainly related to continued reduction of gaseous precursors of secondary $PM_{2.5}$, such as Nitrogen oxides and Sulfur dioxide (e.g. energy production, industry, refineries) and Ammonia (e.g, agriculture), whereas for EC road traffic emissions are expected to be reduced by the introduction of Euro-6. Local measures do not contribute significantly to $PM_{2.5}$ reductions near intense road traffic as direct traffic emissions hardly contribute to $PM_{2.5}$ (Figure 1a). For EC, apart from emission standards such as Euro-6, the introduction of electric vehicles is the most effective measure to reduce EC levels near busy inner-urban roads. However, the impact of the

population-weighted average of this improved air quality near inner-urban roads is limited due to the large number of people living at the urban background.

Only a comparison of the baseline and the 2020 scenarios was considered relevant because no considerable differences in the $PM_{2.5}$ exposure in the three future scenarios are expected (see Table 2).

Table 3

In total 2,097 (Uncertainty Interval (UI): 1,403, 2,711) life years can be saved, by a decrease of 7.41 μg/m^3 $PM_{2.5}$ concentration with marked gender differences for people more than 60 years (see Table 3). Overall, per 1,000 adults 5.8 life years lost can be prevented. More specific contribution to these life years lost can be analyzed for ischemic heart disease and lung cancer. For ischemic heart disease 346 (UI: 269, 413) life years in total could be saved, that is 0.9 YLLs per 1,000 people. For lung cancer mortality a loss of 338 (UI: 138, 504) life years can be prevented, but there are gender differences, and overall per 1,000 people 0.9 YLLs can be prevented.

Table 4

EC is a specific indicator for road traffic emissions and therefore is particularly sensitive for assessment of the impact of traffic measures on local air quality and health impact for people living close to road traffic. Therefore the burden due to EC is only calculated for the people living close to this kind of road, which are in Rotterdam around 3.8% of total population (13,946 people). It has the same exposure patterns for the future scenarios as $PM_{2.5}$. Comparing the 2010 scenario with the BAU scenario results in 0.7 μg/m^3 less EC which could prevent 67 (UI: 46, 98) YLLs equivalent to 7.0 YLLs per 1,000 people.

A comparison of the baseline and the 2020 scenarios result in 262,282 RADs prevented in total by a decrease of $PM_{2.5}$ by 7.41 μg/m^3, with no significant gender disparities (Table not presented). Due to the fact that there is no difference in the modeled $PM_{2.5}$ concentration in the two 2020 scenarios no additional YLLs or RADs can be assigned to those scenarios beyond the benefits of BAU.

The noise levels are expected to increase in 2020 as compared to 2010 due to a growth in motorway traffic of 2% per year up to 2020 (see Table 5). The impact of local measures is limited as these measures are restricted to inner-urban roads which represent about 65% of the total traffic volume in Rotterdam and the remaining 35% of motorway traffic is not affected by local measures. It has to be noted that electric vehicles have no engine noise but still the tyres result in noise emissions.

Table 5

The comparison of the different scenarios by people exposed to noise and the number of people highly annoyed (HA) (Table 6) and highly sleep disturbed (HSD) (Table 7) gives interesting results considering the future traffic scenarios. The lowest burden is expected in the baseline scenario with 550 (UI: 275, 3,300) YLDs due to noise annoyance. In the BAU scenario another 29 (UI: 13, 155) YLDs will be lost due to noise and in scenario a) 7 (UI: 4, 41) and in scenario b) 2 (UI: 0, 9) more YLDs will be lost compared to the baseline. Regarding the future scenarios, scenario b) will cause the lowest loss in life years (552 (UI: 275, 3,309) YLDs) and the BAU the highest with 579 (UI: 288, 3,455) YLDs.

Table 6

The fewest YLDs were lost in the baseline scenarios due to noise sleep disturbance, because a lower number of people were exposed to noise. Compared to the BAU scenario 46 (25, 64) fewer YLDs were lost. The burden in scenario b) is the lowest compared to the other future scenarios, but still higher than the baseline scenario with 4 (UI: 2, 4) more lost life years.

Table 7

By introducing scenario a) compared to the BAU 36 (UI: 20, 51) YLDs can be prevented and even 42 (UI: 23, 60) YLDs by introducing scenario b).

Discussion

The assessed policies were determined by the city itself and the results are therefore policy relevant (Giles et al. 2011). The results for noise implicate an increased burden (YLDs), altough the burden of air pollution from road traffic is supposed to decrease in the future, because of already implemented standards like Euro-6. Therefore noise is becoming an important health risk due to the increasing car fleet. The results should be interpreted carefully due to limitations and uncertainties. Therefore an uncertainty analysis is presented in the following section to give information on the reliability and consistency of the results. The guidance of the *International Programme on Chemical Safety* (IPCS 2008) and from Remais et al. (2014) were followed by doing a qualitative analysis to identify important and crucial factors of uncertainty. Likewise the risk estimates expressed as ERFs were identified as main sources of uncertainty. For that reason, their upper and lower bounds were used to perform a sensitivity analysis (Table 3 and 4). Besides the ERFs the Disability Weights are a very critical part of the HIA, because they are based on people’s judgments. Therefore a lower and upper bound of the Disability Weights were used in the analysis as well (Table 6 and 7).

EC has been used as a sensitive indicator for exhaust emissions from road traffic: in Europe emissions from diesel traffic contribute to 70% of EC concentrations (Bond et al. 2013). In 2020, the levels of EC are expected to decrease due to Euro-6 which is the emissions limit for motor vehicles in Europe since 2014. Due to this BAU development, which will reduce EC levels significantly, the two additional CO_2 reduction measures beyond BAU taken in Rotterdam hardly contribute to further decrease EC levels. It is noted that these conclusions are based on modeling with relative large uncertainties in the input parameters: the traffic volume, the fleet composition and the emission factors for road traffic.

The quantitative effects are rather small, which is in line with other local HIA studies (Joffe and Mindell 2002; Schram-Bijkerk et al. 2009). A direct comparison with other Dutch Environmental Burden of Disease studies, like Hänninen et al. (2014) or HIA studies would be irrelevant, because the results depend highly on the considered policies. Hence a comparison with similar policies and the same reference area would be meaningful, but no such studies were found. When comparing the prevented burden in the BAU scenario with the prevented burden in the future scenarios no effects concerning air pollution and only small effects for noise can be seen. A reason for these results could be the already implemented environmental standards in the city and an expected decrease in air pollution due to Euro-6. The aim of the city of Rotterdam is to increase urban density in the city center (Schaminée et al. 2012) which is most probably correlated with an increase in the number of cars on the motorway around the city, estimated at around 2% per year for motorway traffic and zero for inner-urban roads. Therefore the intervention with the aim of decreasing the use of cars would nearly offset this change. It can be concluded that the policy will have impacts but due to other changes these impacts are rather small.

The comparison of the baseline scenario with the future scenarios also gives some relevant findings. Many life years and RADs can be prevented by a decrease of $PM_{2.5}$ and lower EC levels, which shows that a decrease in air pollution can produce important health benefits and that Rotterdam is proceeding in improving air quality.

The comparison of the baseline scenario with the future scenarios gives negative results for the burden of noise, which means that more healthy life years may be lost due to increasing noise exposure from 2010 to 2020. More people will be exposed to noise because of the assumed general increase of vehicles in Rotterdam.

Population data were gathered from representative national Dutch sources and thus sample uncertainties are probably very small. National morbidity and mortality rates were applied to the population of Rotterdam. Here it is assumed these rates can be applied to the population of Rotterdam. An aspect which needs to be considered concerning the ERFs of $PM_{2.5}$ is that they are only applicable for a population which is exposed over a long period of time. But, there is a high percentage of the population of Rotterdam that is foreign, probably not exposed to Rotterdam's level of $PM_{2.5}$ for their lifetime (Gemeente Rotterdam, Centrum voor Onderzoek en Statistiek 2013). This aspect could not be considered in our assessment, because only the general number of how many people have foreign roots was available and not the duration of their stay in Rotterdam.

Only the concentrations of $PM_{2.5}$, EC and the noise level and their attributed burden of disease in 2010 and 2020 were compared. The years in-between were not considered in detail. Likewise, population changes were not considered, although it could be assumed that there will be an increase in the number of people in Rotterdam. Due to population projections the population will increase up to 640,215 people in 2020 (Hoppesteyn 2012). The expected growth is due to

internal and international migration as well as natural growth. Furthermore age-specific changes in the population structure caused by migration or changes in disease patterns were not considered. Including a population change into the calculations is a complex task and for further assessment it is worthwhile to consider a step-wise approach, because a reduction in air pollution can have an impact on future patterns of death by decreasing the mortality risk and thereby the age-specific death rates. A reduction in air pollution postpones death, which in turn leads to an increase of survival time and a slight increase in the size and age of the population (COMEAP 2010). A life table approach, especially for long-term effects and long latencies can be used.

It is worth remembering that the burden of disease calculated in this impact assessment is an underestimate of the total burden. The list of health impacts that can be calculated do not cover the whole health conditions of the population of Rotterdam. For example, for some health outcomes only adults are considered. Measures of exposure for recognized negative environmental conditions are also an indicator of health of the population.

We calculated the overall burden attributable to $PM_{2.5}$, EC and noise for the population of Rotterdam. However the burden can be much higher when considering also vulnerable subpopulations. Thus analyses for children and elderly people, cyclists and pedestrians would be appropriate, which was not possible due to data limitations (Schaminée et al. 2012).

Conclusions

This study assessed the effects of GHG policies in Rotterdam by quantifying the impacts on health that might be expected from taking some specific transport measures.

The local traffic measures to reduce CO_2 emissions from road traffic concern road traffic on inner-urban roads but *not* on motorways, as the latter are regulated by national authorities. Inner-urban road traffic represents about 65% of the total vehicle kilometers driven within Rotterdam.

Hence Rotterdam only has partial control over reducing CO_2 emissions from road traffic within its boundaries and thus consequently the impact on air quality and related health effects. In Rotterdam (and the rest of Europe), it is expected that the regional background of air pollutants in 2020 will be lower than in 2010 due to further application of cleaner technology. The air quality development of the city of Rotterdam is progressing well, but benefits of emission reduction measures may be partially offset by the increase of vehicle kilometers. Therefore it is worthwhile to consider besides the regulatory measures community and individual interventions (Giles et al. 2011).

It is assumed that the noise levels in Rotterdam will increase due to 2% more cars each year. Thus noise exposure will remain an important aspect to be considered for Rotterdam. Additional interventions to minimize noise could be considered, such as silent asphalt for roads.

While the estimated impacts on health are rather small, they can nonetheless be used to inform policy makers on the co-benefits of the two policies which were developed with the primary aim of reducing GHG emissions. The policies were determined by the city of Rotterdam and are therefore policy relevant and the results can be readily applied in the decision making process.

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Tables

Table 1: Selected air quality and noise indicators and their exposure response function for several health outcomes

Health outcomes (specific population)	ICDX	Causal relationship	Source
Due to $PM_{2.5}$			
All-cause mortality excl. accidents (30 years and older)	A00-R99	1.062 (95% CI 1.040, 1.083) per 10 $\mu g/m^3$	Hoek et al. (2013)
Mortality due to ischemic heart disease (30 years and older)	I20-I25	1.152 (95% CI 1.111, 1.196) per 10 $\mu g/m^3$	Krewski et al. (2009)
Lung cancer related mortality (30 years and older)	C33	1.09 (95% CI 1.04, 1.14) per 10 $\mu g/m^3$	Hamra et al. (2014)
RADs, Restricted activity days (18-64 years)		4.75% (95% CI 4.17, 5.33) per 10 $\mu g/m^3$	Hurley et al. (2005)
Due to EC			
All-cause mortality excl. accidents (30 years and older)	A00-R99	1.06 (95% CI 1.04, 1.09) EC per 1 $\mu g/ m^3$	Janssen et al. (2011)
Due to Noise			
Annoyance	-	$\%HA = 9.868*10^{-4} (L_{den}-42)^3 - 1.436*10^{-2} (L_{den}-42)^2 + 0.5118 (L_{den}-42)$	Miedema and Oudhoorn (2001)
Sleep disturbance	G47	$\%HSD = 20.8 – 1.05 * L_{night} + 0.01486 * L_{night}^2$	Miedema and Oudshoorn (2003)

Noise: Percentage and number of adults annoyed and highly annoyed, indoor, in 1 year. Percentage and number of adults sleep disturbed and highly sleep disturbed, indoor, in 1 year.

Table 2: $PM_{2.5}$ and EC ($\mu g/m^3$) in different scenarios and distributed in locations in Rotterdam

Scenarios \ Location	background		motorway		street		urban		total city[a]	
	$PM_{2.5}$	EC	$PM_{2.5}$	EC	$PM_{2.5}$	EC	$PM_{2.5}$	EC	$PM_{2.5}$	EC
Baseline 2010	14.9	1.1	15.5	1.4	15.8	1.5	15.2	1.2	15.2	1.2
BAU 2020	8.9	0.5	9.2	0.5	9.4	0.8	9.1	0.5	9.1	0.5
Scenario a) 10% less traffic	8.9	0.5	9.2	0.5	9.2	0.7	9.1	0.5	9.1	0.5
Scenario b) 50% electric cars	8.9	0.4	9.2	0.5	9.2	0.6	9.1	0.5	9.1	0.5

[a] population weighted average

Table 3: YLLs prevented due to decrease of $PM_{2.5}$ by 7.41 µg/m^3 from 2010 to 2020 by different health outcomes, in brackets is the confidence limit

Age	Population		Total mortality excl. accidents (ICD-10 A00-R99) (UI)		Ischemic heart disease mortality (ICD-10 I20-I25) (UI)		Lung cancer mortality (ICD-10 C33-C34) (UI)	
	Men	**Women**	**Men**	**Women**	**Men**	**Women**	**Men**	**Women**
30-34	23,192	22,448	14 (9, 18)	13 (9, 17)	2 (2, 3)	1 (1, 1)	0 (0, 0)	0 (0, 0)
35-39	22,672	21,736	25 (17, 32)	22 (14, 28)	5 (4, 6)	1 (1, 1)	1 (1, 2)	1 (1, 2)
40-44	22,142	20,963	36 (24, 47)	33 (22, 43)	9 (7, 10)	2 (1, 2)	3 (1, 4)	4 (2, 6)
45-49	21,182	20,953	52 (35, 68)	52 (35, 68)	10 (8, 13)	5 (4, 5)	7 (3, 10)	9 (4, 13)
50-54	18,656	18,810	75 (50, 97)	68 (45, 87)	17 (13, 20)	7 (5, 8)	12 (6, 18)	15 (7, 22)
55-59	16,421	16,643	97 (65, 125)	81 (54, 105)	19 (15, 23)	7 (5, 8)	20 (10, 29)	20 (9, 28)
60-64	15,873	15,931	120 (80, 155)	95 (63, 122)	28 (21, 33)	9 (7, 11)	27 (13, 39)	21 (10, 30)
65-69	11,281	11,761	112 (75, 145)	84 (56, 109)	23 (18, 28)	11 (8, 13)	27 (13, 39)	17 (8, 24)
70-74	8,930	10,405	120 (80, 155)	99 (66, 128)	23 (18, 28)	14 (11, 17)	23 (11, 34)	15 (7, 21)
75-79	6,926	9,581	129 (86, 167)	128 (85, 165)	27 (21, 32)	18 (14, 21)	21 (10, 30)	12 (6, 17)
80-84	4,805	8,198	114 (76, 147)	154 (103, 199)	23 (18, 28)	24 (19, 29)	13 (7, 19)	7 (3, 10)
85+	3,308	9,261	104 (70, 135)	270 (181, 349)	22 (17, 26)	39 (31, 47)	5 (3, 8)	4 (2, 5)
Total	175,388	186,690	998 (668, 1,291)	1,099 (735, 1,420)	208 (162, 250)	138 (107, 163)	159 (78, 232)	125 (60, 178)

Table 4: YLLs prevented due to a decrease of EC by 0.7 μg/m^3 on major roads from 2010 to 2020, in brackets is the confidence limit

Age groups in years	**Population living close to road traffic 2010 (3.8 % of total population)**		**YLLs for total mortality excl. accidents (UI)**	
	Men	**Women**	**Men**	**Women**
30-34	892	864	0 (0, 1)	0 (0, 1)
35-39	872	836	1 (1, 1)	1 (0, 1)
40-44	852	806	1 (1, 2)	1 (1, 2)
45-49	815	806	2 (1, 2)	2 (1, 2)
50-54	718	724	2 (2, 4)	2 (1, 3)
55-59	632	640	3 (2, 5)	3 (2, 4)
60-64	611	613	4 (3, 6)	3 (2, 4)
65-69	434	452	4 (2, 5)	3 (2, 4)
70-74	344	400	4 (3, 6)	3 (2, 5)
75-79	266	369	4 (3, 6)	4 (3, 6)
80-84	185	315	4 (2, 5)	5 (3, 7)
85+	127	356	3 (2, 5)	9 (6, 13)
Total	6,764	7,182	32 (22, 47)	35 (24, 51)

Table 5: Noise exposed population (in %) grouped by exposure level and scenarios in Rotterdam

L_{den} / L_{night} in decibels	**Baseline 2010**		**BAU 2020**		**Scenario a) 10% less traffic**		**Scenario b) 50 % electric cars**	
	L_{den}	L_{night}	L_{den}	L_{night}	L_{den}	L_{night}	L_{den}	L_{night}
>=49.5 - <54.5	38.8	64.5	38.2	62.4	38.7	64.5	39.0	64.2
>=54.5 - <59.5	30.5	29.1	30.2	30.6	30.6	29.1	30.6	29.2
>=59.5 - <64.5	21.0	5.9	21.0	6.4	20.9	5.9	20.7	6.0
>=64.5 - <69.5	8.3	0.5	9.0	0.6	8.4	0.5	8.2	0.5
>=69.5 - <74.5	1.3	0.1	1.5	0.1	1.4	0.1	1.3	0.1
>=74.5	0.1	-	0.1	-	0.1	-	0.1	-
Total	100	100	100	100	100	100	100	100
Number of people	312,214	109,085	323,356	116,272	315,788	110,909	314,003	109,562

Table 6: Comparison of highly annoyed people and YLDs due to noise exposure (Disability Weight= 0.02) by different scenarios

Exposure to L_{den} in dB(A)	**Baseline 2010**		**BAU 2020**		**Scenario a) 10% less traffic**		**Scenario b) 50% electric cars**	
	People HA	**YLDs**	**People HA**	**YLDs**	**People HA**	**YLDs**	**People HA**	**YLDs**
≥49.5 - <54.5	5,650	113 (57, 678)	5,763	115 (58, 692)	5,705	114 (57, 685)	5,715	114 (57, 686)
≥54.5 - <59.5	7,397	148 (74, 889)	7,605	152 (76, 913)	7,509	150 (75, 901)	7,479	150 (75, 897)
≥59.5 - <64.5	8,129	163 (81, 975)	8,398	168 (84, 1008)	8,172	163 (82, 980)	8,042	161 (80, 965)
≥64.5 - <69.5	5,003	100 (50, 600)	5,584	112 (56, 670)	5,076	102 (51, 609)	4,984	100 (50, 598)
≥69.5 - <74.5	1,201	24 (12, 144)	1,432	29 (14, 172)	1,250	25 (13, 150)	1,228	25 (12, 147)
≥74.5	115	2 (1, 14)	137	3 (1, 16)	134	3 (1, 16)	133	3 (1, 16)
Total	27,495	550 (275, 3,300)	28,920	579 (288, 3,455)	27,847	557 (297, 3,341)	27,580	553 (275, 3,309)

For YLDs in bracket Disability Weights (0.01, 0.12)

Table 7: Comparison of highly sleep disturbed people and YLDs due to noise exposure (Disability Weight= 0.07) by different scenarios

Exposure to L_{night} in dB(A)	**Baseline 2010**		**BAU 2020**		**Scenario a) 10% less traffic**		**Scenario b) 50% electric cars**	
	People HSD	**YLDs**	**People HSD**	**YLDs**	**People HSD**	**YLDs**	**People HSD**	**YLDs**
≥49.5 - <54.5	4,491	314 (180, 449)	4,629	324 (185, 463)	4,564	320 (183, 456)	4,491	314 (180, 449)
≥54.5 - <59.5	2,926	205 (117, 293)	3,285	230 (131, 329)	2,974	208 (119, 297)	2,953	207 (118, 295)
≥59.5 - <64.5	826	58 (33, 83)	952	67 (38, 95)	835	58 (33, 84)	839	59 (34, 84)
≥64.5 - <69.5	89	6 (4, 9)	111	8 (5, 11)	99	7 (4, 10)	100	7 (4, 10)
Total	8,331	583 (334, 834)	8,977	629 (359, 898)	8,472	593 (339, 847)	8,382	587 (336, 838)

For YLDs in brackets Disability Weights (0.04, 0.1)

Figure 1

Modeled exposure data for air quality and noise

a) Annual average $PM_{2.5}$ ($\mu g/m^3$) in Rotterdam 2010
b) Annual average EC ($\mu g/m^3$) in Rotterdam 2010
c) Annual average L_{den} (dB) in Rotterdam 2010
d) Annual average L_{night} (dB) in Rotterdam 2010

11-Feb-2015

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