Journal of Transport Geography 41 (2014) 1–9

Contents lists available at ScienceDirect

Journal of Transport Geography

journal homepage: www.elsevier.com/locate/jtrangeo

Greenhouse gas emissions from flying can offset the gain from reduced driving in dense urban areas

Juudit Ottelin [a,*], Jukka Heinonen [a,b], Seppo Junnila [a]

[a] Aalto University School of Engineering, P.O. Box 15800, 00076 Aalto, Finland
[b] University of Iceland School of Engineering and Natural Sciences, VRII, Hjardarhagi 2-6, 107 Reykjavík, Iceland

ARTICLE INFO

Keywords:
Air travel
Private driving
Greenhouse gas emissions
Urban structure
Travel behaviour
Lifestyles

ABSTRACT

Numerous studies have illustrated how denser urban forms lead to smaller greenhouse gas (GHG) emissions from passenger transport. Many of these studies have excluded aviation since the association between urban structure and air travel is not as intuitive as it is the case of ground travel. However, several recent studies have concluded that air travel is a significant contributor to the GHGs from passenger transport. Furthermore, even air travel habits depend heavily on lifestyles and socio-economic factors that are related to the urban form. Here we analyse the interactions between urban structure and different transportation modes and their GHG impacts in Finland. The study utilises the data from the Finnish Transportation Agency's passenger traffic survey from May 2010 to May 2011, which includes over 12 000 people and over 35 000 trips. The survey is based on one-day travel diaries and also includes additional data on long-distance trips from a longer period. Methodologically, the study takes a traveller's perspective to assess the GHG emissions from passenger transport. We found that (1) air travel breaks the pattern where GHG emissions decrease with increasing density of urban structures, and (2) in the metropolitan region there is a clear trade-off between car-ownership and air travel in the middle income class. The main policy implication of our study is that air travel must be included in GHG assessments and mitigation strategies targeting travel behaviour. In dense urban regions, the emissions of air travel have the potential to offset the gain from reduced private driving.

1. Introduction

Passenger transport is recognised as one of the major causes of greenhouse gas (GHG) emissions and has received wide attention from researchers and policy makers striving for sustainability. Numerous studies have illustrated how denser urban forms lead to smaller GHG emissions (e.g. Newman and Kenworthy, 1989, 1999; Mindali et al., 2004; Norman et al., 2006, see also review by Badoe and Miller (2000)). This has led to GHG mitigation strategies that emphasise densification as an important measure to reduce GHG emissions from transport. However, many of these older studies have excluded aviation and especially the interactions between aviation and other forms of transport, whereas several more recent studies have concluded that the main contributors to the climate change related to passenger transport are private driving and air travel (e.g. Brand and Boardman, 2008; Aamaas et al., 2013; Åkerman et al., 2012).

Currently, aviation causes about 3.1% of the total GHG emissions in the EU and 1.9% globally, but the emissions are rapidly increasing (Hill et al., 2012). Of these, the greatest amount is due to passenger air travel. According to Brand and Preston (2010), passenger air travel has been increasing by about 6% yearly in the UK since the mid-1970s. Åkerman et al. (2012) reported an average rate of increase of 5.5% per year for international passenger air travel during 1980–2007 in Sweden. It has also been predicted, that the share of air travel will increase in the future because the emissions from other sectors are declining, but in the aviation sector the rapid growth of passenger numbers easily overrides any reductions from the technological development (e.g. Hill et al., 2012; Åkerman, 2011). Furthermore, the emissions from air travel have a higher impact on radiative forcing than the emissions from ground transport, though there are some uncertainties involved (e.g. Lee et al., 2010). Moreover, Scott et al. (2010) emphasised that the global tourism sector is unlikely to achieve its share of the GHG emissions reductions targets, mainly due to the emissions from air travel.

The association between urban structure and air travel is not as intuitive as it is in the case of ground travel, but air travel habits do depend heavily on lifestyles and socio-economic factors that are

* Corresponding author. Tel.: +358 50 3752506
E-mail addresses: juudit.ottelin@aalto.fi (J. Ottelin), jukka.heinonen@aalto.fi (J. Heinonen), seppo.junnila@aalto.fi (S. Junnila).

http://dx.doi.org/10.1016/j.jtrangeo.2014.08.004
0966-6923/

related to the urban form. It has been suggested that lifestyles are not only a product of individual's values and personal identity, but that lifestyles, for example consumption and time use, are constrained by the surrounding structure (e.g. Heinonen et al., 2013a,b, Baiocchi et al., 2010; Jalas, 2002). Brand and Preston (2010) found in their case study of Oxfordshire, UK, that when socio-economic, lifestyle or other such factors are not accounted for, the GHG emissions from air travel increase along with the level of urbanisation so that the emissions are lowest in rural and highest in large urban regions. In their study, the increase in the emissions from air travel was found to be significantly higher than the decrease in the emissions from private driving, and thus the total emissions from personal travel were the highest in large urban regions.

Viewed from the perspective of consumption research, e.g. Ornetzeder et al. (2008) have shown that not owning a car may lead to increased consumption in other consumption categories, e.g. holiday travel. Heinonen et al. (2013a,b) have presented similar findings. This interesting issue has been given little attention in the field of transport research and was the main inspiration for this study. People who give up a car may consider that as an environmentally friendly choice. However, e.g. Hares et al. (2010) and Miller et al. (2010) showed in their focus group studies that tourists are often unaware of the climate change impact of holiday travel. Davison and Ryley (2010) found in their case study from East Midlands, UK, that only 8% of travellers were trying to reduce air travel for environmental reasons, whereas for example price sensitivity had much more effect. Some studies (e.g. Davison et al., 2014; Barr et al., 2010) have also highlighted that particularly for air travel there is a cognitive dissonance between attitudes and behaviour, i.e. those who recognise the environmental impacts of aviation actually fly more than average and possibly justify this to themselves by environmental friendly choices in other areas of life (Dickinson et al., 2010). Moreover, Frändberg and Vilhelmson (2011) found that even though there is a trend of reducing everyday travel and car-dependency among young Swedish adults, the international long-distance travel of these same people is increasing. They suggested that this is due to their more globalised lifestyles. Furthermore, Holz-Rau et al. (2014) found that while socioeconomic issues affect long-distance trips and daily trips much the same way, the urban form affects mostly in different directions. They found that residents of low-density neighbourhoods make less and shorter long-distance trips than those living in large urban areas and high-density neighbourhoods, but the latter travel less in their daily lives.

In this study, we analyse the interactions between urban structure and different transport modes and their GHG impacts in Finland. The particular aim is to investigate the role of aviation in the composition of the total GHG emissions from passenger transport in different urban structures. The analysis is based on descriptive statistics and mean value comparisons. The study depicts how urban form, household characteristics, travel behaviour and GHG emissions from transport are interconnected and how, depending on the situation, flying can act as a substitute or as a complement to private driving. The study utilises the data from the Finnish Transportation Agency's passenger traffic survey from May 2010 to May 2011. That data includes one-day travel diaries of 12 000 people and additional information about long-distance trips during 2–4 weeks depending on the transport mode.

This paper contributes to the literature by providing a new case study that simultaneously studies daily trips and long-distance travel from a perspective of sustainability (here GHG emissions) and includes a measure for urban structure. Our results give support to the earlier findings of e.g. Brand and Preston (2010), who found that the emissions of air travel are highest in large urban regions. Our results also show that the emissions of air travel differ between urban forms within Helsinki Metropolitan Region (HMR). Furthermore, we found a clear trade-off between car-owning and holiday air travel in the middle income class in HMR.

The paper begins by presenting the research material, data processing and method of analysis. In the next section, we show the results of our analysis: (1) How the profile of GHG emissions from personal travel varies in different urban structures and (2) How the profile differs in motorized and non-motorized, i.e. car-owning and car-free, households. In the discussion section, we interpret our results, analyse the uncertainties and give some policy implications. The paper ends with a short conclusion.

2. Research design

2.1. Research material

The main data source of the study is the latest National Travel Survey from May 2010 to May 2011 (Finnish Transport Agency, 2012). The traditional survey is conducted by the Finnish Transportation Agency every six years, and it gives an overall picture of passenger transport in Finland. The survey is based on one-day travel diaries and phone-interviews of over 12 000 people and includes over 35 000 trips. There is detailed information about the trips, such as travel distance, destination and modes of transport as well as demographic information about the respondent. The survey was executed so that the test days varied among the respondents, and the whole survey covers every day of the year including weekends and holidays. In addition to the one-day travel diaries, additional data on long-distance trips was collected. The respondents reported their over 100 km car trips during two weeks before the test day and their over 100 km trips with other transport modes during four weeks before the test day.

In practice, the data is divided into three datasets: background information, one-day travel diaries and long-distance trips. We utilised all three datasets in our study. The same background information was combined with the one-day diary dataset and long-distance trips dataset. It should be noted that about 17.6% of the respondents did not travel at all – they did not even walk during the test day – according to the travel diaries and phone interviews. Still, these respondents are included in the background information dataset and in this study. This is a relatively high amount: e.g. Madre et al. (2007) concluded in their review about immobility in travel surveys that the share of immobiles should be around 8–12% for the standard one-day, weekday only travel diary. However, the Finnish National Travel Survey includes also weekends and holidays. Furthermore, the seasons affect travel behaviour in Finland so that people travel less in winter. If only spring and autumn weekdays are included, the immobility is about 12.7%, according to the survey. The National Travel Survey provides also analytic weights to correct the biases in the demographics of the sample, and we employed these in our study.

We utilised the same research material in our earlier conference paper (Ottelin and Heinonen, 2014), which was presented in the CIB International Conference on Construction in a Changing World, 4th–7th May 2014 in Sri Lanka. In the conference paper, we presented some preliminary and unrefined results of our study.

2.2. Data processing

2.2.1. One-day travel diaries and long-distance trips

For private driving and public transport we utilised both one-day travel data and data on long-distance trips. To avoid double counting, we excluded over 100-km-long trips from the one-day travel diary dataset. Flights and boat trips, of which there are very few observations in the one-day diary dataset, are much less frequent than car trips and trips on public transport. Thus we used only the data on long-distance trips for these two transport modes, even though it means that ⩽100 km boat trips and flights are left out.

2.2.2. Geographical settings

We studied separately Helsinki Metropolitan Region (HMR) and the rest of Finland, since HMR has unique features and represents a notable portion (25%) of the Finnish population. Also, the main airport of Finland is located in Vantaa in HMR. HMR stands for the Helsinki-Espoo-Vantaa metropolitan area and a commuter belt of 15 smaller municipalities. There are about 1.4 million inhabitants in HMR, and the density ranges from 2830 inhabitants/km^2 in Helsinki to 90 inhabitants/km^2 in the commuter belt. Espoo and Vantaa together fall in between these with their 840 inhabitants/km^2. The whole population of Finland is about 5.4 million, and its density is 17.5 inhabitants/km^2.

2.2.3. Measure of urban structure

There are many possible measures to describe urban structure and its density. We utilised a relatively crude measure of density as the main indicator for describing urban form: an areal efficiency ratio e_a created in the Finnish Environment Institute (FEI) and provided in the National Travel Survey. Areal efficiency ratio is similar to a more common floor area ratio but covers a larger area than the building site. It is defined as

$$e_a = \text{floor space}/\text{area}.$$

The FEI indicator has a grid of 250 m * 250 m squares that covers whole Finland. For each square, the value of e_a is the mean of the nine adjacent squares. Thus the e_a-ratio represents the mean efficiency of a wider area.

In the National Travel Survey there are five areal efficiency classes. To increase the sample sizes and the statistical significance of the results we combined some of these. Also, since we were mostly interested in the metropolitan area, the limit of the highest areal efficiency class $e_a > 0.32$ was insufficient for our purposes. Thus we included also one accessibility zone, the pedestrian zone, to illustrate the densest areas. Accessibility zones, also created by FEI, are more loosely defined than the efficiency ratio e_a. The pedestrian zone is the innermost zone of the city, with maximum accessibility to public transport system and commercial centre. In general we preferred the areal efficiency over the accessibility zones because it is a more transparent measure and also showed clearer differences especially in the GHG emissions from private driving. We gave the efficiency classes descriptive names: sparse, low-rise and high-rise. High-rise class, however, includes also some mixed low- and high-rise areas but is predominantly high-rise.

2.2.4. Comparison between non-motorized and motorized subgroups

One of the aims of the study was to test a working hypothesis that there may be a trade-off between private driving and air travel, as suggested by earlier literature (e.g. Ornetzeder et al., 2008; Heinonen et al., 2013a,b). To explore this, we divided our data in motorized and non-motorized categories, i.e. car-owning and car-free households. However, the average income in the car-free households is substantially lower than in the car-owning households. Since we suspected that the trade-off may be based on rebound effects of consumption, i.e. money saved from not owning a car is spent elsewhere, we divided our data further in three income classes to get more meaningful comparisons. We utilised gross income per capita, i.e. the gross household income divided by household members, children included. To avoid comparing families, which are usually motorized, and more often non-motorized single- and couple-households, we also kept these two subgroups separated.

To test the statistical significance of the difference in mean GHG emissions from air travel in the motorized and non-motorized subgroups, we employed the Wilcoxon-Mann–Whitney-test (Wilcoxon, 1945; Mann and Whitney, 1947), which is a non-parametric rank-sum test that can be used for testing the equality of means. We could not use the t-test, since the GHG emissions from air travel are not normally distributed in our data.

2.2.5. Sample sizes

Tables 1 and 2 show the sample sizes of the studied groups. Table 1 also shows the average household size in the urban form classes, and our data supports the general finding pointing to decrease in household size with increasing density of urban structures.

2.2.6. Level of motorisation in different urban forms

Families with children possess a car more often than households without children as can be seen in Tables 2 and 3. Table 3 gives the level of motorisation for different urban forms. A clear majority of families have a car in their household even in the densest urban forms, whereas in the singles and couples subgroup the level of motorisation decreases drastically as the urban structures become denser.

2.2.7. Greenhouse gas coefficients

To calculate the GHG emissions we used a simple screening assessment that includes vehicle operation phase emissions but not the life cycle emissions from manufacture of vehicles or fuels. However, we made an effort to improve the accuracy of the calculation for private driving and air travel, since these two modes make up the majority of the emissions. Short flights create substantially more emissions per flight kilometre than long-distance flights, because the highest amount of emissions is released during the ascent (Chapman, 2007). We employed travel-distance based GHG coefficients for air travel. As for private driving, the emissions are related to the driving speed and number of stops. Because of traffic congestion, traffic lights and pedestrian crossings driving is less efficient in population centres than in highways. We developed a model to take this into consideration. Of course, also the plane and car type affect the emissions, but these could not be taken into account in this study due to data restrictions. However, even though car type significantly affects the emissions on household level, the effect attenuates when larger groups are studied. This issue is further analysed in the uncertainties Section 4.2.

The study relies on the so-called LIPASTO study by the Technical Research Centre of Finland VTT (2012) as the main source of the GHG coefficients. The LIPASTO study includes a vast amount of empirical information about the emissions of different modes of transport in Finland.

The LIPASTO study gives three different GHG coefficients for private driving according to the driving style. Street driving is defined as typical driving in urban areas where the average speed is 30 km/h and where there are one to three stops per kilometre. The GHG coefficient of street driving is 214 CO_2-eq g/km. For highway driving, it is 141 CO_2-eq g/km with the average speed of 95 km/h. The LIPASTO study gives an average GHG coefficient of 167 CO_2-eq g/km for all private driving. It is based on the assumption that the share of street driving is 35% and of highway driving 65%, which is derived from empirical data in Finland.

We assumed that the long-distance car trips of over 100 km are mainly driven on highway and thus employed the GHG coefficient of highway driving for these. For the ⩽100 km car trips we created a model to define the GHG coefficient. The one-day travel diary data includes information about the areal efficiency of the starting and ending point of the trip. We utilised this information and also took into account the timing of the trip, i.e. rush hours in the urban

Table 1
The urban form classes of the study and corresponding sample sizes.

Urban form	Areal efficiency	Sample size			Proportion	Average
	e_a	Singles and couples	Families	Total	(weighted)[a] (%)	Household size
Helsinki Metropolitan Region (HMR)						
Sparse	<0.02	90	107	197	6.7	3.4
Low-rise	0.02–0.16	461	608	1069	35.0	3.1
High-rise	>0.16	922	533	1455	48.7	2.3
Pedestrian zone		193	70	263	9.6	2.2
Finland excluding HMR						
Sparse	<0.02	1406	1269	2675	30.5	3.0
Low-rise	0.02–0.16	2839	1980	4819	51.0	2.7
High-rise	>0.16	750	236	986	9.7	2.0
Pedestrian zone		717	135	852	8.9	1.8

[a] Analytic weight included in the National Travel Survey to correct the demographic biases.

Table 2
The sample sizes of the studied motorized and non-motorized subgroups. The subgroups with a sample size <10 (in parenthesis below) were excluded from the study.

Income per capita (€/year)	Singles and couples		Families	
	Non-mot.	Motorized	Non-mot.	Motorized
Helsinki Metropolitan Region (HMR)				
<15 000	119	87	37	247
15–30 000	166	337	23	486
>30 000	138	513	(5)	182
Finland excl. HMR				
<15 000	459	871	50	1297
15–30 000	286	1794	(6)	1243
>30 000	68	1158	(0)	196

areas. The trips starting and ending in dense areas and driven during the rush hours where given the LIPASTO study's GHG coefficient of street driving, i.e. 214 CO_2-eq g/km. Trips starting and ending in sparse areas and also all trips having an average speed >85 km/h were given the GHG coefficient of highway driving, 141 CO_2-eq g/km. For the purpose of the study, we also calculated two average GHG coefficients based on the coefficients provided in the LIPASTO study: (214 + 167)/2 = 191 CO_2-eq g/km and (141 + 167) = 154 CO_2-eq. Trips that only started or ended in dense areas were given the average GHG coefficient of 191 CO_2-eq g/km if the trip was driven during rush hours and the average GHG coefficient of 154 CO_2-eq g/km if not. Because of the high amount of missing information, however, most of the trips were still given the LIPASTO study's average coefficient of all driving, i.e. 167 CO_2-eq g/km. The effect of the model on the average GHG emissions of residents in different urban forms was relatively small, ranging from −5% to +5%, compared to the case where all the trips were given the same coefficient (167 CO_2-eq g/km).

For public transport, air travel and boat travel we utilised the GHG coefficients provided by the LIPASTO study and shown in Table 4 below. For boat travel, we used the GHG coefficient of a car ferry travelling between Finland and Sweden at 18 knot speed, representing the main ferry type and ferry route in Finland.

It should be noted that the CO_2 equivalents of the LIPASTO study include only CO_2, CH_4 and N_2O emissions which are changed into the CO_2 equivalents according to the Kyoto Protocol (GWP_{100}, global warming potential with a 100-year time horizon). For ground travel, this is an accurate estimate of the climate impact, but in the case of aviation this leads to an underestimation. The climate impact of aviation consists of the long-term impacts from CO_2 and shorter-term impacts from non-CO_2 emissions, including water vapour and NO_x (Lee et al., 2010). There is an ongoing debate about the total effect of aviation on radiative forcing. According to Lee et al. (2010) especially the effect of aircraft-induced cloudiness is still very uncertain. The authors suggested aviation impact multipliers ranging from 1.3 to 2.0 for GWP_{100} and from 2.1 to 4.8 for GWP_{20} (GWP with a 20-year time horizon). However, they also highlighted that the level of scientific understanding of this issue is still low. To be cautious, we employed no aviation impact multiplier in this study, which should be taken into consideration when interpreting the results.

We calculated the GHG emissions for each trip by multiplying the travel distance by the GHG coefficient determined by the mode of transport. The GHG emissions of each car trip were divided by the number of passengers in the car (the National Travel Survey includes this information for each trip). Other GHG coefficients were directly in the form of g/passenger-km. Then we summed the total GHG emissions from each transport mode for each respondent. With the resulting data we were able assess the mean GHG emissions per capita of chosen groups.

2.3. Method of analysis

There are two main methods for assessing the environmental impacts caused by a city or other given area: the consumption-based method and the production-based method. Both have been utilised in transport research. Here we use the consumption-based method to calculate GHG emissions caused by passenger transport. The consumption-based method has a traveller's perspective. GHG emissions caused by residents of a given area are calculated based on the actual travelling of the residents. All the trips are included regardless of where they take place, including trips to secondary

Table 3
The proportion of residents having one or more cars in their household.

Urban form	e_a	HMR		Finland excluding HMR	
		Singles and couples (%)	Families (%)	Singles and couples (%)	Families (%)
Sparse	<0.02	89	100	91	99
Low-rise	0.02–0.16	83	98	80	98
High-rise	>0.16	55	84	65	90
Pedestrian zone		54	90	54	88

Table 4
The GHG coefficients of transport modes (Technical Research Centre of Finland VTT, 2012).

Public transport Mode	CO2e (g/pkm)[a]	Air travel Flight distance	CO2e (g/pkm)	Private driving Driving type	CO2e (g/km)	Boat travel Mode	CO2e (g/pkm)
Train	22	<463 km	260	Highway	141	Ferry	223
Bus, long-distance	43	463–1000 km	178	Street	214		
Bus, local	36	1000–3000 km	149	Average driving	167		
Metro	11	>3000 km	114	Average rush hour[b]	191		
Tram	54			Average non-rush hour[b]	154		

[a] CO2-eq g/passenger-km.
[b] The average coefficients calculated for the purpose of the study, not provided by VTT.

and holiday homes and vacations abroad. This is in contrast to the production-based method, which is employed to assess the environmental impacts from a given geographical area so that all the impacts originating within the area are included regardless of the cause. Thus, in the case of transport, through-traffic is allocated to the area where it occurs. In the production-based method, air travel is normally left out, because it is not meaningful to calculate the emissions from overflights and no city or region is willing to take the responsibility of the emissions from an airport. The consumption-based method is ideal for allocating GHG emissions from air travel to the cause, though also a hybrid-method has been suggested for regional purposes (Wood et al., 2010).

Business trips are excluded in the consumption-based method, since they are correctly allocated to the enterprise rather than the individual. Commuters are included however, because one can choose the mode of transport and affect the distance by choosing the residential location. The consumption-based method enables comparisons between groups, for example between residents of different urban forms. It can also be used to explore the trade-offs between transport modes and more generally the rebound effects of consumption (Wiedenhofer et al., 2013; Heinonen et al., 2013b).

2.4. Further issues of concern

2.4.1. About self-selection bias

Even though a vast number of studies have demonstrated an association between surrounding urban structure and travel behaviour, association alone does not prove causality. There is an ongoing debate about the effect and magnitude of residential self-selection (see reviews by Cao et al., 2009; Mokhtarian and Cao, 2008; Ewing and Cervero, 2010). Socioeconomic factors, attitudes, lifestyles and preferences towards transport modes affect the choice of residence. Therefore, when we study the residents of a specific urban form, the sample is not randomly chosen but self-selected. In statistics, this is referred to as self-selection bias. For example, transit-oriented neighbourhoods attract people who do not own a car. If the reason for not owning a car is socioeconomic or attitudinal, it is not the built environment that is causing the car-free lifestyle but rather the car-free lifestyle that is determining the place of residence.

Cao et al. (2009) reviewed 38 empirical studies dealing with the connection between built environment and travel behaviour, and addressing the self-selection problem somehow. They concluded that there is resounding evidence that built environment has a distinct influence on travel behaviour after self-selection is accounted for. However, they also concluded that residential self-selection substantially attenuates the influence of built environment. Furthermore, the quantitative amounts of the impacts of built environment and self-selection are mainly unknown. In two of the reviewed studies, the impact of residential self-selection was stronger than the impact of built environment (Bagley and Mokhtarian, 2002; Kitamura et al., 1997), and, in eight studies, built environment had the stronger impact (e.g. Schwanen and Mokhtarian, 2005; Salon, 2006). Three studies quantitatively estimated the proportion of the impact of built environment on travel behaviour, and these estimates range from 52% (Salon, 2006) to 90% (Zhou and Kockelman, 2008). Cao et al. concluded that studies that do not take self-selection into account may lead to misleading results and inefficient or even flawed policies.

We appreciate the previous studies analysing the effects of self-selection. In our study, we were unable to take self-selection properly into account because of the restrictions of our data. We included only one aspect of self-selection, family structures, and studied separately singles and couples and larger families. However, our aim was not to define how urban structure affects travel behaviour but rather to show how a different understanding about the sustainability of different urban structures arises when the air travel habits are taken into account. We analyse the effect of self-selection on our results further in the discussion Section 4.1.

3. Results

The main findings of the study are that (1) air travel indeed breaks the pattern of decreasing GHG emissions with increasing density of urban structures and contributes significantly to the total GHG emissions from personal travel in dense urban forms and (2) in HMR there is a clear trade-off between car-ownership and air travel in the middle income class. The results are analysed further in the following sections and illustrated in Figs. 1 and 2.

3.1. GHG emissions in different urban forms

Fig. 1 gives a general view of the GHG emissions from passenger transport in Finland. It shows that the total emissions do not follow the urban structure consistently but that the patterns differ between the studied subgroups. On average, the metropolitan dwellers have higher overall emissions: lower ground transport emissions and significantly higher emissions from air travel. Within all subgroups, the emissions from travel tend to be slightly higher in looser urban structures; however, dense pedestrian zones seem to be an interesting exception, having a slightly upward trend in transport emissions. Most of the total GHG emissions come from private driving and air travel – boat travel and public transport play only a minor role.

3.2. Trade-off between private driving and air travel

Fig. 2 depicts the difference in GHG emissions from transport in motorized and non-motorized subgroups in HMR (Fig. 2a) and elsewhere in Finland (Fig. 2b). Non-motorized households in general are the source of fewer emissions than motorized households. However, emissions from air travel are higher in the non-motorized households in virtually all subgroups. This trade-off between car-ownership and air travel is most distinctive in the middle income class in HMR, where the emissions caused by motorized and non-motorized households come very close to each other.

We tested the statistical significance of the results with the Wilcoxon–Mann–Whitney-test. The null-hypothesis was that

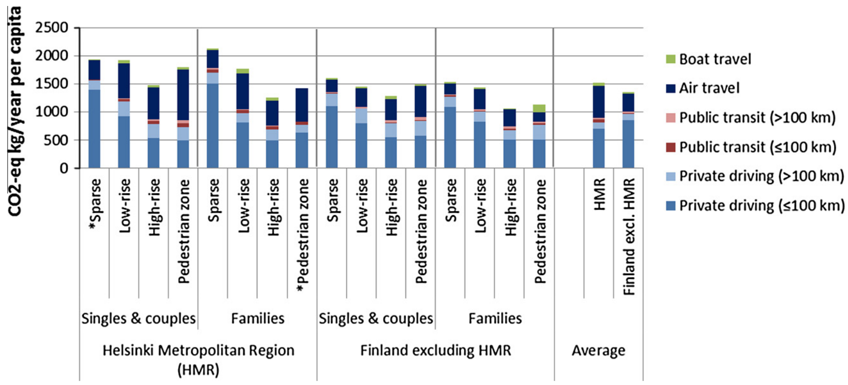

Fig. 1. GHG emissions from passenger transport per capita in different urban forms in subgroups of singles and couples and larger families in HMR and elsewhere in Finland. [*] Sample size <100.

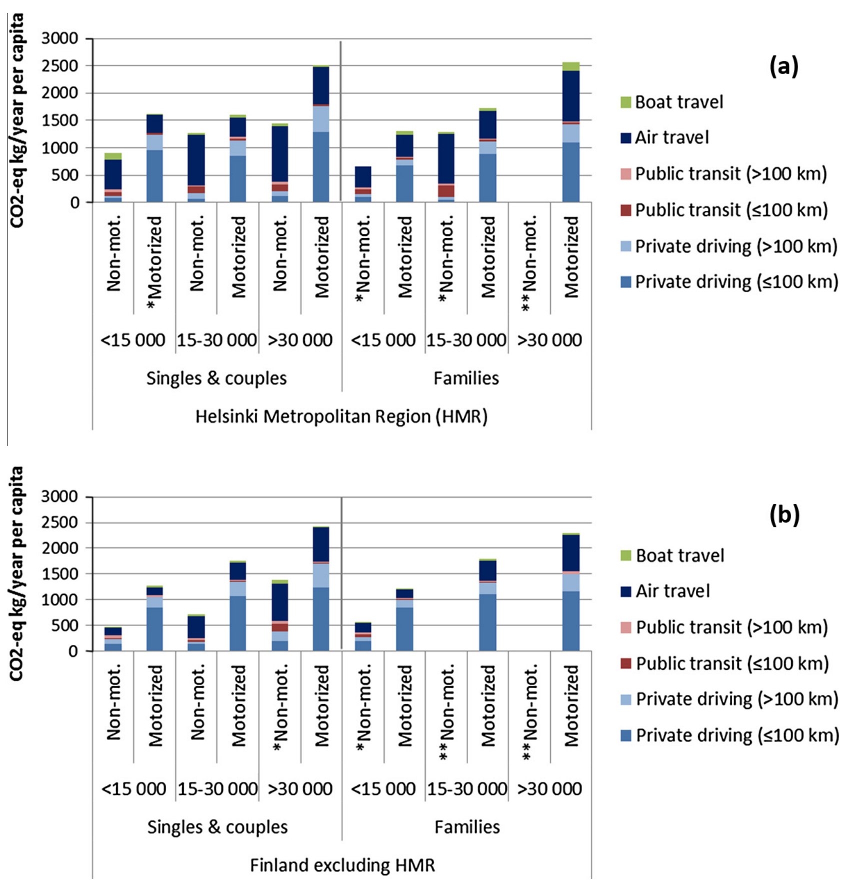

Fig. 2. GHG emissions from transport per capita in the motorized and non-motorized subgroups in different income classes in HMR (a.) and elsewhere in Finland (b.). [*] Sample size <100, [**] Sample size <10, excluded from the study.

GHG emissions from flying do not differ in non-motorized and motorized households. The null-hypothesis could only be rejected in one subgroup: the middle income class singles and couples in HMR with 98.1% significance level ($p = 0.019$). In this subgroup, also the qualitative analysis showed the clearest difference.

4. Discussion

4.1. Interpretation of results

The main results of our study are that (1) air travel breaks the pattern of decreasing GHG emissions with increasing density of urban forms and contributes significantly to total GHG emissions from personal travel in dense urban areas and (2) in Helsinki Metropolitan Region (HMR) there is a clear trade-off between car-ownership and air travel in the middle income class.

In the study, we analysed the interconnections between urban form, household characteristics, travel behaviour and GHG emissions from personal travel in Finland. The emphasis of our study is on air travel, since it has often been neglected in studies discussing the association between built environment and travel behaviour. Even though air travel may not have a direct connection to the urban form, it is connected to the socioeconomic factors, attitudes and lifestyles which are related to the urban form. Furthermore, previous studies have shown that the effect of residential self-selection is strong also in the case of other transport modes (see reviews by Cao et al., 2009; Mokhtarian and Cao, 2008; Ewing and Cervero, 2010). Cao et al. (2009) concluded that policies based on studies that do not take self-selection into account are likely to be inefficient or even flawed. Yet urban density has become one of the key measures in the attempts to mitigate the emissions from private driving.

We believe that excluding air travel from the mitigation strategies also leads to inefficient policies. It is not known whether reducing private driving would lead to changes in travel behaviour in general. Our results imply that there is a trade-off between private driving and air travel. We do not know the mechanism behind the trade-off – it may be based on a simple rebound-effect of consumption, i.e. money saved from not owning a car is used elsewhere, on holiday flights in our case (Ornetzeder et al., 2008; Heinonen et al., 2013a,b). It seems also plausible that car-free households are less likely to make domestic holiday trips by car and may thus spend their holidays more often abroad. Frändberg and Vilhelmson (2011) found in their case study from Sweden that while car-dependency and domestic leisure travel among young adults have decreased, international travel by them has at the same time increased substantially. They also pointed out that the lengthening of the post-adolescent period and postponement of familial and professional obligations leaves more freedom for young adults to travel internationally.

In conclusion, the observed differences in the GHG profiles of personal travel in different urban forms may actually be more strongly associated to differing lifestyles than the physical features of the built environment. Since urban density has nevertheless become an important tool to reduce private driving, more detailed studies on how this affects the overall travel behaviour, including international travel, should be conducted. Even a small change in air travel may be significant, since the emissions of one return flight from Finland to Thailand equal the emissions of private driving per person per year. Also, it is likely that the true climate impact of flying is stronger than the CO_2, CH_4 and N_2O emissions alone indicate. Lee et al. (2010) suggested aviation impact multipliers ranging from 1.3 to 4.8. We did not employ any multiplier for the GHG emissions from air travel, and thus our results are likely to be underestimates. For the sake of comparison, Brand and Preston (2010) utilised an aviation impact multiplier of 3.0 for flights over 700 km. Had we employed a similar multiplier, the weight of air travel would have increased substantially, of course. For example, in the middle income class in HMR, the total emissions from transport in the non-motorized subgroup would have been higher than in the motorized subgroup and close to equal in the two other income classes in HMR. Furthermore, the role of air travel is likely to grow in the future, since the current technological development in combustion technologies and alternative fuels of cars is fast whereas in the field of aviation similar technological leaps are not in sight, especially when keeping in mind the rapid growth of passenger kilometres and trips (Åkerman, 2011; Scott et al., 2010; Chapman, 2007).

It has been also pointed out that a behavioural change in travel habits, especially reduction in car use, is hard to achieve and political interventions are often unsuccessful (Graham-Rowe et al., 2011; Steg and Gifford, 2005), possibly because there are latent attitudes and deeply-rooted preferences regarding transport modes (e.g. Buys and Miller, 2011; Vij et al., 2013). Therefore, it might be more effective to focus the mitigation measures on technological development. However, many researchers have stressed the need for both behavioural and technological changes to tackle the GHG emissions of transport, and these two approaches do not need to be seen as contradictory (e.g. Bastani et al., 2012; Ewing and Cervero, 2010; Chapman, 2007). Brand and Preston (2010) suggest carbon pricing (carbon tax) or downstream cap-and-trading (personal carbon allowances and a trading system). They also suggest that the emissions from personal travel and domestic energy use should be integrated to a personal profile covering all GHG emissions. This approach would be ideal from the equality point of view, since it would leave the consumer the possibility to make personal choices about where to cut the emissions. For example, a person living in a suburban area, owning a low-carbon hybrid or electric car and rarely flying can cause less emission than a person living in a small apartment in a dense urban area but flying once or twice a year.

4.2. Main uncertainties and suggestions for further study

The study involves uncertainties related to three areas: the travel survey data, the GHG coefficients and the method. Even though one-day travel diaries are a customary way to study travel behaviour, it is based on the assumption that the one day on which respondents have been administered the diary is representative of their general travel behaviour, which clearly does not always hold true. Also, people may forget some of their trips or be dishonest. However, the large sample sizes in the studied groups reduce the uncertainties related to the travel diary data. Nonetheless, in some studied groups the amount of respondents is too low to be entirely representative. In our figures, we have marked the sample sizes below 100 with a star (∗). In addition, some information is missing, especially about the income level, possibly because that kind of information is regarded somewhat sensitive – about 20% of respondents left it blank. This may also bias the results a little.

However, considering that the study focuses on air travel, the greatest flaw in the travel survey data is the small number of people who flew within the asked four weeks: only 4.5% of all respondents and 7.3% in HMR. Because of this, the mean emissions from air travel are much more uncertain than the emissions from e.g. private driving. In future, it would be important to lengthen the reporting period for long-distance trips in order to get richer data. Also, some new important variables could be included, like ticket payer and detailed trip purpose.

The GHG coefficients employed in the study are simplified. Particularly, we could not take into account the vehicle or plane type, which affect the emissions substantially. In the case of private driving, e.g. Brownstone and Golob (2009) showed in their case study from California that the fleet in dense urban areas is generally newer and more energy efficient than in rural areas. However, the Finnish Transport Safety Agency's statistics about registered cars (currently in traffic use) show that the differences between cities are quite small in Finland. For example, the mean emissions of passenger cars are 166.2 CO_2 g/km in HMR and 170.5 CO_2 g/km elsewhere in Finland. We did not utilise this information in our study, because it was not compatible with the GHG coefficients of the Technical Research Centre of Finland VTT (2012).

In the case of air travel, the CO_2 equivalents used in the study exclude important short-lived GHG emissions, like water vapour and NO_x, which contribute to the climate impact of flying (Lee et al., 2010). Furthermore, there is an ongoing debate concerning especially the effect of aircraft-induced cloudiness (Lee et al., 2010). To be cautious, we did not employ any multiplier for the GHG emissions from air travel, and thus our results are likely to underestimate the true climate impact of flying.

The model we created to take into account the effect of driving style, i.e. street or highway driving, also has limitations. We had only secondary data about the driving style, i.e. the average speed, time of day and information about the urban form of the start- and end point of the trip. Another problem was that most of the trips had missing information in at least one of these categories. Despite of its weaknesses, the model worked in the expected direction: increase in the GHG emissions from private driving in the densest urban forms and decrease in the emissions in sparsely populated areas. However, the effect was quite small. In the future, it would be important to get a better idea of the magnitude of the effect of the driving style on the GHG emissions from rural vs. urban areas.

It would also be of high importance to consider the whole lifecycle of transport systems in the assessments of GHG emissions. Chester et al. (2013) and Chester and Horvath (2009) have already illustrated how the infrastructure and production chains of vehicles, fuels and energy increase GHG emissions from each transport mode. In their general example concerning the US, Chester and Hovarth showed that, compared to pure tailpipe emissions, there was an increase of 63% for onroad, 155% for rail traffic and 31% for aviation in the GHG emissions from the whole transport system. Furthermore, the contribution of vehicle production and infrastructure rises even more if the time-correction of the emissions is considered (Kendall and Price, 2012). GHG emissions occurring at the dawn of a product life cycle have a higher global warming impact than those occurring later.

Moreover, attributional assessment is insufficient because of the rapid development in the transport field. Bastani et al. (2012) showed in their study how the uncertainty of future technology and transport needs affect considerably the expected value of the total fuel consumption and GHG emissions from the fleet. Thus a consequential approach would be more informative for policy implications. Also, it would be interesting to explore urban areas other than HMR, because the characteristics of a city have their own effect to the results. For example Li et al. (2010) demonstrated, in their study of Chinese megacities, different reasons for car ownership in different cities.

5. Conclusions

The contribution of the study to the literature of transport research is twofold: (1) it is a new case study that simultaneously studies daily trips and long-distance travel from a perspective of sustainability (here GHG emissions) and includes a measure for urban structure. Our results give support to the earlier findings of e.g. Brand and Preston (2010), who found that the emissions of air travel are highest in large urban regions. (2) It demonstrates a clear trade-off between car-owning and holiday air travel in the middle income class in Helsinki Metropolitan Region, which gives support to the earlier findings in the field of consumption research (e.g. Ornetzeder et al., 2008; Heinonen et al., 2013a,b).

Air travel has often been neglected in studies of built environment and travel behaviour. Even though the association between urban structure and air travel is not as intuitive as it is in the case of ground travel, air travel habits do depend heavily on lifestyles and socio-economic factors that are related to the urban form. Furthermore, previous studies have shown that also in the case of other transport modes, travel behaviour is strongly affected by residential self-selection (see reviews by Cao et al., 2009; Mokhtarian and Cao, 2008; Ewing and Cervero, 2010). Residential self-selection means that, among other things, attitudes, lifestyles and travel preferences affect the choice of residence.

Cao et al., 2009 concluded that policies based on studies that do not take self-selection into account are likely to be inefficient. We believe that the continuing concentration on private driving and exclusion of air travel also leads to inefficient policies. The interconnections between urban form, lifestyles and travel behaviour are complicated, and it is not known whether reducing private driving would lead to changes in travel behaviour and lifestyles in general. However, there are only a few studies so far that would include rich data about both long-distance travel and daily short-distance travel, let alone other aspects of lifestyles such as living habits. Thus, further studies on the subject are warranted.

The main policy implication of our study is that GHG assessments and mitigation strategies aimed to reduce GHG emissions from passenger transport, e.g. by changing the built environment, should include air travel. Brand and Preston (2010) suggested carbon pricing (carbon tax) or downstream cap-and-trading (personal carbon allowances and a trading system). They also recommended that emissions from personal travel and domestic energy use should be integrated to a personal profile covering all GHG emissions. This approach would be ideal from the equality point of view, since it would leave the consumer the possibility to make personal choices about where to cut the emissions.

Acknowledgements

The authors thank the following organizations for the funding that has enabled this research: Energizing Urban Ecosystems (EUE) program, city of Espoo, Fortum and Academy of Finland Grant No. 140938. The views expressed by the authors do not necessarily reflect those of the funders.

References

Aamaas, B., Borken-Kleefeld, J., Peters, G.P., 2013. The climate impact of travel behavior: A German case study with illustrative mitigation options, Environmental Science & Policy, vol. 33, November 2013, pp. 273–282.
Åkerman, J., 2011, Transport systems meeting long-term climate targets: a backcasting approach, KTH Royal Institute of Technology. Doctoral Dissertation.
Åkerman, J., 2012. Climate impact of international travel by Swedish residents. J. Transp. Geogr. 25 (November), 87–93.
Badoe, D.A., Miller, E.J., 2000. Transportation–land-use interaction: empirical findings in North America, and their implications for modeling. Transport. Res. Part D Transp. Environ. 5 (4), 235–263.
Bagley, M.N., Mokhtarian, P.L., 2002. The impact of residential neighborhood type on travel behavior: a structural equations modeling approach. Ann. Reg. Sci. 36 (2), 279–297.
Baiocchi, C., Minx, J., Hubacek, K., 2010. The impact of social factors and consumer behavior on carbon dioxide emissions in the United Kingdom. A regression based on input–output and geodemographic consumer segmentation data. J. Ind. Ecol. 14, 50–72.
Barr, S., Shaw, G., Coles, T., Prillwitz, J., 2010. 'A holiday is a holiday': practicing sustainability, home and away. J. Transp. Geogr. 18 (3), 474–481.
Bastani, P., Heywood, J.B., Hope, C., 2012. The effect of uncertainty on US transport-related GHG emissions and fuel consumption out to 2050. Transport. Res. Part A: Policy Pract. 46 (3), 517–548.
Brand, C., Boardman, B., 2008. Taming of the few—the unequal distribution of greenhouse gas emissions from personal travel in the UK. Energy Policy 36 (1), 224–238.
Brand, C., Preston, J.M., 2010. '60-20 emission'—The unequal distribution of greenhouse gas emissions from personal, non-business travel in the UK. Transp. Policy 17 (1), 9–19.
Brownstone, D., Golob, T.F., 2009. The impact of residential density on vehicle usage and energy consumption. J. Urban Econ. 65 (1), 91–98.
Buys, L., Miller, E., 2011. Conceptualising convenience: transportation practices and perceptions of inner-urban high density residents in Brisbane, Australia. Transp. Policy 18 (1), 289–297.
Cao, X., Mokhtarian, P.L., Handy, S.L., 2009. Examining the impacts of residential self-selection on travel behaviour: a focus on empirical findings. Transp. Rev. 29 (3), 359–395.
Chapman, L., 2007. Transport and climate change: a review. J. Transp. Geogr. 15 (5), 354–367.
Chester, M., Horvath, A., 2009. Environmental assessment of passenger transportation should include infrastructure and supply chains. Environ. Res. Lett. 4 (2).
Chester, M., Pincetl, S., Elizabeth, Z., Eisenstein, W., Matute, J., 2013. Infrastructure and automobile shifts: positioning transit to reduce life-cycle environmental impacts for urban sustainability goals. Environ. Res. Lett. 8 (1).
Davison, L., Ryley, T., 2010. Tourism destination preferences of low-cost airline users in the East Midlands. J. Transp. Geogr. 18 (3), 458–465.
Davison, L., Littleford, C., Ryley, T., 2014. Air travel attitudes and behaviours: the development of environment-based segments. J. Air Transp. Manage. 36 (April), 13–22.
Dickinson, J.E., Robbins, D., Lumsdon, L., 2010. Holiday travel discourses and climate change. J. Transp. Geogr. 18 (3), 482–489.
Ewing, R., Cervero, R., 2010. Travel and the built environment. J. Am. Plann. Assoc. 76 (3), 265–294.
Finnish Transport Agency, 2012. National Travel Survey 2010–2011.
Frändberg, L., Vilhelmson, B., 2011. More or less travel: personal mobility trends in the Swedish population focusing gender and cohort. J. Transp. Geogr. 19 (6), 1235–1244.

Graham-Rowe, E., Skippon, S., Gardner, B., Abraham, C., 2011. Can we reduce car use and, if so, how? A review of available evidence. Transport. Res. Part A: Policy Pract. 45 (5), 401–418.
Hares, A., Dickinson, J., Wilkes, K., 2010. Climate change and the air travel decisions of UK tourists. J. Transp. Geogr. 18 (3), 466–473.
Heinonen, J., Jalas, M., Juntunen, J.K., Ala-Mantila, S., Junnila, S., 2013a. Situated lifestyles: I. How lifestyles change along with the level of urbanization and what the greenhouse gas implications are—a study of Finland. Environ. Res. Lett. 8 (2).
Heinonen, J., Jalas, M., Juntunen, J.K., Ala-Mantila, S., Junnila, S., 2013b. Situated lifestyles: II. The impacts of urban density, housing type and motorization on the greenhouse gas emissions of the middle income consumers in Finland. Environ. Res. Lett. 8 (3).
Hill, N., Brannigan, C., Smokers, R., Schroten, A., van Essen, H., Skinner, I., 2012. Developing a better understanding of the secondary impacts and key sensitivities for the decarbonisation of the EU's transport sector by 2050. European Commission and AEA Technology plc. <http://www.eutransportghg2050.eu> (accessed 11.12.13).
Holz-Rau, C., Scheiner, J., Sicks, K., 2014. Travel distances in daily travel and long-distance travel: what role is played by urban form? Environ. Plann. A 46 (2), 488–507.
Jalas, M., 2002. A time use perspective on the materials intensity of consumption. Ecol. Econ. 41, 109–123.
Kendall, A., Price, L., 2012. Incorporating time-corrected life cycle greenhouse gas emissions in vehicle regulations. Environ. Sci. Technol. 46 (5), 2557–2563.
Kitamura, R., Mokhtarian, P.L., Laidet, L., 1997. A micro-analysis of land use and travel in five neighborhoods in the San Francisco Bay Area. Transportation 24 (2), 125–158.
Lee, D.S., Pitari, G., Grewe, V., Gierens, K., Penner, J.E., Petzold, A., Prather, M.J., Schumann, U., Bais, A., Berntsen, T., Iachetti, D., Lim, L.L., Sausen, R., 2010. Transport impacts on atmosphere and climate: aviation. Atmos. Environ. 44 (37), 4678–4734.
Li, J., Walker, J., Srinivasan, S., Anderson, W., 2010. Modeling private car ownership in China. Transport. Res. Rec. J. Transport. Res. Board 2193, 76–84.
Madre, J., Axhausen, K.W., Brög, W., 2007. Immobility in travel diary surveys. Transportation 34 (1), 107–128.
Mann, H.B., Whitney, D.R., 1947. On a test of whether one of two random variables is stochastically larger than the other. Ann. Math. Stat. 18 (1), 50–60.
Miller, G., Rathouse, K., Scarles, C., Holmes, K., Tribe, J., 2010. Public understanding of sustainable tourism. Ann. Tourism Res. 37 (3), 627–645.
Mindali, O., Raveh, A., Salomon, I., 2004. Urban density and energy consumption: a new look at old statistics. Transport. Res. Part A: Policy Pract. 38 (2), 143–162.
Mokhtarian, P.L., Cao, X., 2008. Examining the impacts of residential self-selection on travel behavior: a focus on methodologies. Transport. Res. Part B Meth. 42 (3), 204–228.
Newman, P., Kenworthy, J., 1989. Cities and Automobile Dependence: An International Sourcebook. Gover, Aldershot.
Newman, P., Kenworthy, J., 1999. Sustainability and Cities: Overcoming Automobile Dependence. Island Press, California.
Norman, J., MacLean, H., Kennedy, C., 2006. Comparing high and low residential density: life-cycle analysis of energy use and greenhouse gas emissions. J. Urban Plan. Develop. 132 (1), 10–21.
Ornetzeder, M., Hertwich, E.G., Hubacek, K., Korytarova, K., Haas, W., 2008. The environmental effect of car-free housing: a case in Vienna. Ecol. Econ. 65 (3), 516–530.
Ottelin, J., Heinonen, J., 2014. Supporting sustainable transportation by urban structure development – the role of air travel. In: CIB International Conference on Construction in a Changing World, 4th–7th May 2014, Sri Lanka.
Salon, D., 2006. Cars and the City: An Investigation of Transportation and Residential Location Choices in New York City, Davis, CA: University of California. Doctoral dissertation.
Schwanen, T., Mokhtarian, P.L., 2005. What affects commute mode choice: neighborhood physical structure or preferences toward neighborhoods? J. Transp. Geogr. 13 (1), 83–99.
Scott, D., Peeters, P., Gössling, S., 2010. Can tourism deliver its "aspirational" greenhouse gas emission reduction targets? J. Sustain. Tourism 18 (3), 393–408.
Steg, L., Gifford, R., 2005. Sustainable transportation and quality of life. J. Transp. Geogr. 13 (1), 59–69.
Technical Research Centre of Finland VTT, 2012. LIPASTO – a calculation system for traffic exhaust emissions and energy consumption in Finland. <http://lipasto.vtt.fi> (accessed 09.12.13).
Vij, A., Carrel, A., Walker, J.L., 2013. Incorporating the influence of latent modal preferences on travel mode choice behavior. Transport. Res. Part A: Policy Pract. 54 (August), 164–178.
Wiedenhofer, D., Lenzen, M., Steinberger, J.K., 2013. Energy requirements of consumption: urban form, climatic and socio-economic factors, rebounds and their policy implications. Energy Policy 63 (December), 696–707.
Frank, Wilcoxon, 1945. Individual comparisons by ranking methods. Biometrics Bull. 1 (6), 80–83.
Wood, F.R., Bows, A., Anderson, K., 2010. Apportioning aviation CO_2 emissions to regional administrations for monitoring and target setting. Transp. Policy 17 (4), 206–215.
Zhou, B., Kockelman, K., 2008. Self-selection in home choice: use of treatment effects in evaluating relationship between built environment and travel behavior. Transport. Res. Rec. J. Transport. Res. Board 2077, 54–61.