WP 1: City typologies, climate impact functions and library for urban agglomerations

Task 1.3- Benchmarking of vulnerability, adaptiveness and efficiency of city components

Deliverable 1.3
Methods inventory for infrastructure assessment

Reference code: RAMSES – D1.3

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**Short Description:**
This report presents benchmarks, inventories and quantitative measures to assess infrastructure components, efficiencies and strategies that support the implementation of adaptation and mitigation efforts at the urban level.

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Table of contents

Executive Summary ........................................................................................................................................... v
List of Abbreviations........................................................................................................................................ 1
List of Figures .................................................................................................................................................. 3
List of Tables .................................................................................................................................................. 4
List of Equations ............................................................................................................................................ 5

1   Introduction to benchmark vulnerability, adaptiveness and efficiencies ............................................. 6

2   International, national policies and cities strategies .............................................................................. 10
  2.1  Introduction to climate change policies, strategies and plans............................................................ 10
  2.2  Investigating climate change strategies across 200 EU cities ......................................................... 11
  2.3  Results of national and local policies on city strategies ................................................................. 12
  2.4  Implications and conclusions of investigating policies and strategies ........................................... 17

3   Methods and examples of benchmarking cities ..................................................................................... 20
  3.1  Introduction to benchmarking cities ................................................................................................. 20
  3.2  Contemporary methods for city benchmarking ............................................................................... 21
  3.3  Benchmarking and ranking of 88 European cities .......................................................................... 24
  3.4  Conclusions of benchmarking methods and using DEA ............................................................... 25

4   Cost inventories for 9 infrastructure components .................................................................................. 27
  4.1  Introduction to cost inventories ......................................................................................................... 27
  4.2  Gathering and standardising cost data for 9 infrastructure components ......................................... 28
  4.3  Cost inventories for infrastructure components ............................................................................ 30
  4.4  Implications for RAMSES case study cities and countries ........................................................... 43
  4.5  Conclusions from investigating the infrastructure components .................................................... 45

5   Assessment and selection methods for Green Urban Infrastructures (GUI) ........................................ 47
  5.1  Introduction to streetscape and GUI ................................................................................................. 47
  5.2  Developing the Performance Index (PI) .......................................................................................... 48
  5.3  Testing the Performance Index (PI) ................................................................................................. 51
  5.4  Discussion, future research and conclusions using PI .................................................................... 55

6   Cities strategies and Electric Vehicles (EV) ........................................................................................... 57
  6.1  Introduction to emissions targets, EV and infrastructure ................................................................. 57
  6.2  Investigating mitigation strategies and EV registration ................................................................... 59
  6.3  Results from analysing city strategies and EV uptake to meet targets ........................................... 60
6.4 Discussions of cities strategies and EV incentives .................................................. 65
6.5 Conclusions and implications to meet carbon reduction targets.......................... 65

7  The trinity of scaling to assess cities ....................................................................... 67

7.1 Introduction to scaling and city assessments ....................................................... 67
7.2 Urban CO₂ emission and data collection ............................................................. 68
7.3 City efficiency and emissions ............................................................................. 69
7.4 Theoretical country-wide urban emissions and correlations .............................. 72
7.5 Discussion and conclusions of using scaling........................................................ 74

8  City characteristics and CO₂ efficiency ................................................................. 76

8.1 Introduction for assessing city efficiencies .......................................................... 76
8.2 Data gathering and processing .......................................................................... 77
8.3 Data analysis and influences of population density............................................. 81
8.4 Discussions of population density and emissions ................................................ 85
8.5 Conclusions and policy implication in increasing efficiencies ............................ 86

9  Main conclusions from this report (D1.3) ............................................................. 87

9.1 Main research areas and conclusions ................................................................. 87
9.2 Relationship and relevance of D1.3 to future RAMSES work ............................ 88
9.3 Final conclusions ................................................................................................ 89

References ................................................................................................................. 90
Executive Summary

This report comprises RAMSES Deliverable 1.3 (D1.3), which is the main output from Task 1.3 (T1.3). It presents benchmarks, inventories and quantitative measures to assess infrastructure components, efficiencies and strategies that support the implementation of adaptation and mitigation efforts at the urban level. Building on the research reported in RAMSES D7.1 (Pryck et al., 2014), we explored in T1.3 the relation between the centrality features, the accessibility networks and, the diversity of facilities across RAMSES cities and wide range of countries.

In the Description of Work (DoW) of RAMSES (Kropp, 2013) we state that parts of our research should result in peer reviewed journal papers. Based on the research that we have conducted in T1.3 a wide range of papers are already published, accepted, submitted or are drafted for submission to peer reviewed scientific journals. This report builds on some of the papers as described in the relevant chapters, although research as part of D1.3 is still underway and yet to be reported to the RAMSES network and the wider research community. We present the findings of benchmarking efficiencies, strategies, plans and performances of cities. As stated in the DoW we utilise case studies to assess Green Urban Infrastructures and Electric Vehicles infrastructure systems and provide cost inventories for selected infrastructure components across RAMSES cities and beyond. The work of T1.3 was partly informed and driven by the research conducted and reported by our RAMSES colleagues and the wider RAMSES network, but also has implications to the future research of RAMSES project as discussed in the relevant chapters and summarised below.

Considering the findings reported by RAMSES D7.2 (Domingos et al., 2015) and 8.2 (Mendizabal et al., 2016) we argue that urban climate change mitigation and adaptation implementation strategies require a context-oriented and systems-based approach. To help quantify such approaches we benchmark the local efforts in terms of strategies, socio-economic efficiency and environmental impacts in Chapters 2 and 3. Reporting and benchmarking 11 national and 200 city strategies in Chapter 2 we argue that multiple interests and motivations are inevitable. Defining the influences of national policies on city strategies and plans did inform parts of the benchmarking and efficiency assessment research conducted in T1.3 but also has wider relevance to future RAMSES Tasks 5.4 and 7.3 as well as Work Packages (WP) on stakeholder engagement and city transitions (WP8, 9 and 10). In Chapter 3 we rank 88 European cities using different analytical methods, to demonstrate how city efficiencies and rankings are dependent on the benchmarking methods applied. This research can help in testing alternative transition models as part of RAMSES WP8. It also helps to identify the best adaptation or energy efficiency options based on previous performance, which is helpful in WP10 (training and toolbox).

Building on RAMSES D2.2 (Acero et al., 2014), D2.4 (Kallaos et al., 2015a) and D5.1 (Floater et al., 2014) we gathered and benchmarked the costs of constructing or restructuring infrastructure components that support adaptation and mitigation efforts. In Chapter 4, we provide detailed cost inventories for air conditioning, solar blinds, permeable paving, green roofs, levees, loft insulation, double glazing and solar panels. The inventories - some of the most comprehensive yet published - cover commercial prices across 24 countries in 13 languages including countries with RAMSES case study cities: London, Antwerp, Bilbao, New York, Hyderabad, Bogota, and Rio de Janeiro. We find that RAMSES case study cities with higher Gross Domestic Product (GDP) per capita had the largest amount of data available and argue that transferrable methodologies for economic assessments are needed. Data from the inventories set out here will be used in the estimation of aggregate adaptation costs to be developed in RAMSES D5.3. Furthermore, and beyond the RAMSES project, Chapter 4 represents an effort to provide a set of much needed standardized data that can be used to evaluate different adaptation policies. In order for policy makers to better allocate resources, it becomes necessary to evaluate, compare and prioritize different adaptation options, for which this data is vital.
As described by RAMSES D2.1 (Kallaos et al., 2014), D2.3 (Kallaos et al., 2015b) and D2.4 (Kallaos et al., 2015a) Green Urban Infrastructure (GUI) and Electric Vehicles (EV) are important city components that support adaptation and mitigation efforts. We conducted case studies to provide quantitative assessments for such infrastructure systems and present our key results of two case studies in Chapters 5 and 6. RAMSES D4.3 (Lobaccaro et al., 2016) presented methodologies to develop GUI design recommendations for adaptation i.e. urban heat island effect, and we researched the contributions GUI can make on mitigation. With the Performance Index (PI) we present and demonstrate in Chapter 5 an assessment tool of GUI that was applied in real conditions, which has implications for future RAMSES work such as the transition alternatives testing (Task 8.3) and toolbox on climate change adaptation and sustainability and training (Task 10.1). The tool can help city planners select the most suitable species to maximize the mitigation and adaptation potential of GUI vegetation. In Chapter 6 we investigate how 30 UK Local Authorities provide infrastructures, strategies and incentives to increase the uptake of EV. We find that either the incentives for EV purchase and actual operation seems beyond the abilities of cites or that the climate mitigation policies published by the cities are ineffectual. This has relevance to the ongoing RAMSES research transition (Task 8.3), stakeholder dialogues (T9.2) and the wider policy implications currently researched in Task 7.3.

Considering the research reported in RAMSES D1.2 (Boettle et al., 2016) we used generic approaches in Chapters 7 and 8 to assess and, where feasible, provide quantitative methods to analyse the efficiency of individual city components and technologies to reduce CO₂ emissions (i.e. mitigation) or in terms of its internal resilience (i.e. adaptation). Utilising the urban GHG emission from 256 cities we provide quantitative methods to analyse the efficiency of city structures in Chapter 7. We conclude that urbanisation may drive climate change in developing countries and may mitigate climate change in developed ones, which has a relevance to T8.3. The findings do suggest that density-scaling might play a role encouraging us to study the influence of population density on urban CO₂ emissions in the following Chapter. We describe the relationship between population density and the CO₂ emissions in Chapter 8 for some of the RAMSES cites. We suggest that cities should no longer be viewed in isolation or as bounded entities but, as dynamic temporal and spatial clusters. Cities do evolve in space and time, and our findings have implications to the ongoing research on policy tools in T7.3, transition alternative testing T8.3, and to the stakeholder dialogue as part of T9.2.

To benchmark the vulnerability, adaptiveness and efficiencies of cities, we have developed and present in this report, various inventories to help researchers, urban planners and city representatives to assess individual city strategies, infrastructure components and efficiencies. The research described in this report provides qualitative and quantitative evidence for the complexity of an urban systems and the attributional problems of costs. We have developed and tested various aggregated or reduced forms of cities and infrastructure components, used case studies and identified key elements, methods and strategies. We combined sector-based assessments with regionalised and localised assessments to determine the key climatic impacts and studied certain types of infrastructural components, i.e. physical and social and assessed these in terms of the usefulness, resilience, or by the usage of quantitative measures. This report is intended to promote improved and integrated assessments across temporal and spatial clusters and improves the implementation of infrastructure components that support climate change adaptation and mitigation efforts in urban areas.
List of Abbreviations

ADEME- Agence de l’environnement et de la maîtrise de l’énergie (French environmental agency)
AGB- Aboveground Biomass
AT- Austria
BE- Belgium
BEP- Biomass Energy Potential
BG- Bulgaria
BMU- Bundesministerium fur Umwelt (Federal Environment Ministry in Germany)
BR- Brazil
BVOC- Biogenic Volatile Organic Compounds
Cap- Capita
CCA- City Clustering Algorithm
CH- Switzerland
CIRIA- Construction Industry Research and Information Association
CO- Carbon Monoxide
CO- Colombia
CO2- Carbon Dioxide
CO2e- Carbon Dioxide equivalent
CoM- Covenant of Mayors
CPF- Crown Projection Factor
CSP- Carbon Sequestration Potential
D1.3- Deliverable 1.3
DBH- Diameter at Breast Height
DE- Germany
DEA- Data Envelopment Analysis
DfT- Department for Transport in the UK
DK- Denmark
DoW- Description of Work
DVLA- Driver and Vehicle Licensing Agency in the UK
EC- European Commission
ES- Spain
EST- Environmental Stress Tolerance
EU- European Union
EV- Electric Vehicle(s)
FI- Finland
FP- Framework Programme
FR- France
GDP- Gross Domestic Product
GHG- Greenhouse Gas
GLC- Global Land Cover
GOSAT- Greenhouse gases Observing SATellite
GRUMP- Global Rural-Urban Mapping Project
GUI- Green Urban Infrastructure
HHV- Higher Heating Value
HR- Croatia
HVAC- Heating, Ventilation and Air Conditioning
IAL- Intra-Annual Leaf cover
ICLEI- International Council for Local Environmental Initiatives
IE- Ireland
IMELS- Italian Ministry for the Environment, Land and Sea
IN- India
IPCC - Intergovernmental Panel on Climate Change
IT- Italy
LAI- Leaf Area Index
LHV- Lower Heating Value
MKD- Macedonia
NAS- National Adaptation Strategy
NL- Netherlands
NO2- Nitrogen Dioxide
O3- Ozone
OLS- Ordinary Least Squared
PFP- Pollution Flux Potential
PI- Performance Indicator
PO- Poland
PPS- Power Purchasing Standard
PT- Portugal
RAMSES- Reconciling Adaptation, Mitigation and Sustainable Development for citiES
RO- Romania
ROS- Reactive Oxygen Species
RU- Russia
SE- Sweden
SEAP- Sustainable Energy Action Plans
SET- Strategic Energy Technology
List of Figures

Figure 1-1 Overall structure of Task 1.3 and this report D1.3 ................................................................. 6
Figure 2-1 Snapshot of cities surveyed ........................................................................................................ 13
Figure 2-2 Global, EU and national policies/networks and city climate change plans ......................... 17
Figure 3-1 Urban Audit indicators used to benchmark European cities ................................................. 23
Figure 3-2 Compare ranking of OLS, SFA and DEA for European cities ................................................. 24
Figure 3-3 Pairwise comparison of city rankings using DEA method of ranking ..................................... 25
Figure 4-1 Average air conditioning cost across countries ..................................................................... 31
Figure 4-2 Costs of air conditioning in RAMSES cities ....................................................................... 32
Figure 4-3 Cost to GDP ratio of air conditioning in RAMSES cities ..................................................... 33
Figure 4-4 Cost of air conditioning excluding labour ............................................................................... 33
Figure 4-5 Average costs of green roofing by country ........................................................................... 34
Figure 4-6 Costs of green roofing by type ................................................................................................. 34
Figure 4-7 Average construction costs of levees in selected EU countries ............................................. 35
Figure 4-8 Average cost of mechanical ventilation in selected RAMSES cities .................................. 37
Figure 4-9 Costs of permeable paving by country .................................................................................. 37
Figure 4-10 Average costs and total area (double glazing) ..................................................................... 39
Figure 4-11 Average costs of loft insulation ............................................................................................. 39
Figure 4-12 Costs of loft insulation in RAMSES cities ........................................................................... 40
Figure 4-13 Cost to GDP ratio of loft insulation for selected RAMSES cities ....................................... 40
Figure 4-14 Average costs of solar panels across countries .................................................................... 41
Figure 4-15 Average costs of solar panels in RAMSES cities ............................................................... 41
Figure 4-16 Cost to GDP ratio for solar panels in RAMSES cities ....................................................... 42
Figure 4-17 Total cost vs power in India ................................................................................................ 42
Figure 4-18 Total cost vs power in the USA ............................................................................................ 43
Figure 4-19 GDP and data available in the case study cities (adaptation) ........................................... 45
Figure 5-1 Site extent (image taken from Google Maps) ........................................................................ 51
Figure 6-1 Selection of UK cities analysed .......................................................................................... 59
Figure 6-2 Climate change mitigation measures (mentioned by the 28 cities) ................................. 61
Figure 6-3 Level of carbon reduction within the private vehicle sector ............................................. 64
Figure 7-1 City efficiency in terms of CO₂ emissions ........................................................................... 69
Figure 7-2 Economic development and city efficiency in terms of CO2 emissions .............................. 70
Figure 7-3 Exponents of the power-law relating population and area from meta-study ................. 71
Figure 7-4 Illustration of theoretical country-wide urban emissions ............................................... 72
Figure 7-5 Correlations of country population and population of its largest city. ............................... 73
Figure 7-6 Gasoline use/capita and population density ......................................................................... 74
Figure 8-1 Application of City Clustering Algorithm ........................................................................... 79
Figure 8-2 Differences between GRUMP and GLC data ................................................................. 81
Figure 8-3 Population density and total emissions (per capita) at 5km threshold distance ............. 82
Figure 8-4 Population density, buildings and on-road emissions per capita ..................................... 84

List of Tables
Table 1-1 Published, submitted and drafted papers for peer reviewed journals from T1.3 ................. 7
Table 2-1 National, city plans and Covenant of Mayors (CoM) across 11 countries ............................ 15
Table 4-1 Summary of search terminologies and languages for cost inventory research ................ 29
Table 4-2 Summary of data collected for cost inventories .................................................................... 30
Table 4-3 Installation costs of air conditioning in EU and non-EU countries .................................... 31
Table 4-4 Costs of levees ..................................................................................................................... 36
Table 4-5 Average costs of mechanical ventilation ................................................................................ 36
Table 4-6 Costs of permeable paving by type ....................................................................................... 37
Table 4-7 Average costs of solar blinds (USA) .................................................................................... 38
Table 4-8 Average costs of double glazing ............................................................................................ 38
Table 4-9 Average costs of loft insulation by type .................................................................................. 40
Table 4-10 Costs of infrastructure components for heat waves in RAMSES cities ............................. 43
Table 4-11 Costs of infrastructure components for flooding in RAMSES cities ................................. 44
Table 4-12 Costs of infrastructure components supporting mitigation efforts in RAMSES cities .... 44
Table 5-1 GUI and street vegetation species used in evaluation ............................................................ 52
Table 5-2 Parameters and methods to estimate multi-functionality and resilience traits ................ 53
Table 5-3 Calculation of Performance Index (PI) .................................................................................. 55
Table 6-1 UK Urban Audit cities strategies and electric cars registered ............................................. 62
Table 6-2 Wilcoxon- test evaluating effectiveness of urban EV strategies ......................................... 63
Table 6-3 Ultra-low emission vehicles registration Jan 2010 to Dec 2014................................. 63
Table 8-1 Methods used to assess city density and CO₂ emissions............................................. 80
Table 8-2 Linear fitting of cluster distance .................................................................................... 82
Table 8-3 Cluster population and emissions .................................................................................. 83
Table 8-4 Top 5 urban clusters in terms of emissions................................................................... 85
Table 9-1 Research summary of D1.3.......................................................................................... 87

List of Equations

Equation 3-1 City efficiency ........................................................................................................ 21
Equation 3-2 City inefficiency ..................................................................................................... 21
Equation 3-3 City inefficiency using SFA approach................................................................. 21
Equation 3-4 Weighted efficiencies of a city ............................................................................. 22
Equation 3-5 Output oriented VRS efficiency across cities.................................................... 22
Equation 4-1 Cost to GDP ratio ................................................................................................ 32
Equation 5-1 Pollution Flux Potential (PFP) ............................................................................. 49
Equation 5-2 Carbon sequestration potential (CSP)................................................................. 49
Equation 5-3 Higher Heating Value (HHV) .............................................................................. 50
Equation 5-4 Bio Energy Potential (BEP) ................................................................................ 50
Equation 5-5 Environmental stress tolerance .......................................................................... 50
Equation 5-6 Crown protection factor ..................................................................................... 51
Equation 8-1 Emissions per capita and the population density ................................................ 82
1 Introduction to benchmark vulnerability, adaptiveness and efficiencies

It is widely accepted that anthropogenic greenhouse gas emissions are higher than ever and have increased since the pre-industrial era, which is largely due to economic and population growth (IPCC, 2015). The continued emission of greenhouse gases will cause further warming and long-lasting changes in people’s lives and the ecosystems as a whole. The risk of climate-related impacts and increased vulnerability and exposure of human and natural systems requires efforts on both fronts climate change mitigation i.e. addressing the causes of climate change and climate change adaptation i.e. dealing with the consequences of climate change. Without adaptation, risks of economic losses from extreme events are substantial in cities and there are significant co-benefits, synergies and trade-offs exist between mitigation and adaptation, in particular in urban areas (IPCC, 2015). The Paris Agreement within the framework of the United Nations Framework Convention on Climate Change (UNFCCC, 2015) aims to govern greenhouse gases emissions mitigation, adaptation and finances. 195 countries agreed to reduce emissions as part of the method for reducing greenhouse gas. Also climate change adaptation and financing options were discussed but no agreement was reached yet.

To feed into the process RAMSES can provide some much needed quantified evidence on the causes and impacts of climate change and the costs and benefits of a wide range of adaptation options, focusing on cities. This report comprises the RAMSES Deliverable 1.3 (D1.3), which is the main output from Task 1.3 (T1.3). The report presents benchmarks, inventories and quantitative methods for assessing city components, efficiencies and strategies that support the implementation of adaptation and mitigation efforts.

Based on DoW of RAMSES (Kropp, 2013) the overall structure of Task 1.3 and this Deliverable (D1.3) is illustrated in Figure 1-1, which can be summarised as follows. To benchmark the vulnerability, adaptiveness and efficiencies of cities, the aim of T1.3 and this Report (D1.3) is to develop various methods and inventories that help assess city measures, components and strategies. The focus is on adaptation, mitigation and costs to support the development of urban strategies and implementation of infrastructure components. As suggested by the DoW we utilised a variety of research methods that were entitled in the DoW as benchmarking, case studies and generic approaches. We present our research results in individual chapters as illustrated in Figure 1-1.

![Figure 1-1 Overall structure of Task 1.3 and this report D1.3](image-url)
As part of its strategic relevance and the potential impact described in the DoW (Kropp, 2013) RAMSES will improve the integration of adaptation research into decision-making as “there is a need to assess cities regarding their plans and actions to achieve efficiency in reducing climate related impacts and greenhouse gas emissions.” (p.57). As explicitly stated in DoW Work Package 1 of RAMSES (Kropp, 2013), parts of our research should result in peer reviewed journal papers. Based on the research conducted in T1.3 a range of papers have already being accepted, published, submitted or are in draft form for submission to peer reviewed scientific journals. All the papers listed below already have (or will) acknowledge the RAMSES project and the funding body EC 7th FP (contract Reference 308497). The papers are listed by their short title alphabetically in Table 1-1, which are shorter titles of the full (or draft) titles in order to provide a sense of the research that was covered by T1.3. Some of the papers are building the basis for the Chapters of this report (D1.3). Our research (T1.3) and this report (D1.3) has greatly benefited from a range of RAMSES activities and outputs such as previous RAMSES reports, discussion with RAMSES consortium partners and the interactions amongst the various RAMSES Work Packages as described within the relevant chapters.

Table 1-1 Published, submitted and drafted papers for peer reviewed journals from T1.3

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<tr>
<td>Building stock model across Europe</td>
<td>Submitted</td>
<td>(Sandberg et al., Under review)</td>
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<tr>
<td>City as a nuclei of sustainability</td>
<td>In Press</td>
<td>(Rybski et al., In Press)</td>
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<td>City density and CO₂</td>
<td>Published</td>
<td>(Gudipudi et al., 2016)</td>
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<tr>
<td>EU strategies and the impact on city plans</td>
<td>Published</td>
<td>(Heidrich et al., 2016)</td>
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<tr>
<td>Benchmarking Green Urban Infrastructures</td>
<td>Published</td>
<td>(Tiwary et al., 2016)</td>
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<tr>
<td>Key technologies and strategies for London</td>
<td>Submitted</td>
<td>(Villarroel Walker et al., Under review)</td>
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<tr>
<td>The LAYERS of supplies for city structures</td>
<td>Drafted</td>
<td>(Heidrich et al., Draft)</td>
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<td>Uncertainty analysis for estimating the costs of sea dykes</td>
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<td>(Lenk et al., Draft)</td>
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<tr>
<td>Urban mitigation strategies and EV uptakes</td>
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<td>(Heidrich, Draft)</td>
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Conducting such diverse research within T1.3 but also across RAMSES as a whole, may create misunderstandings and issues of definition that could limit the transferability of research findings across RAMSES, the various T1.3 studies and the following chapters. Where needed, we defined terminologies that are specific to the research questions and provide caveats for direct comparison or inference across the research reported in the chapters e.g. city, efficiency and benchmark. The terminologies and analytical focuses of cities, efficiencies and benchmarking are further described in the chapters, but for clarity we discuss them briefly as follows.

We conducted research across cities from different continents, countries, at various scales and levels of analysis. Until recently, a harmonized and consistent definition of cities that allowed comparisons and ability to conduct cross-country research of cities did not exist (Dijkstra and Poelman, 2012). Cities, also called Urban Areas, are defined by administrative and/or political boundaries and we provide explanations of the scale of analysis in the relevant chapters.

Efficiency is a quantifiable measure of output relative to input. It avoids wasting materials, energy, efforts, money, and time in doing something or in producing a desired result (Bousoffiane et al., 1991). Efficiency can be expressed in various ways; for example efficiency of work, pollution per capita, fuel efficiency, socio-economic and regulatory efficiencies and many more. We defined the efficiency that we investigate within each chapter.

Benchmarking, in turn, is the comparison of processes and performance metrics to identify best practices. It can be a technique that is characterized by the systematic search for procedures and solutions for complicated problems and processes in order to achieve improvements. In city research benchmarking uses indexes, assessments and surveys to help track trends, economic and investment patterns, and the effects of political or regulatory changes across cities (Global Benchmarking Network, 2015). Within our research and the chapters we expand the definitions and make them case specific and relevant to our research and the analysis in hand.
As was decided by WP1 and described in the 18 month interim report (Kropp, 2014) part of the research conducted by RAMSES focuses on specific case study cities. The RAMSES cities are (in alphabetical order): Antwerp (the Netherlands), Bilbao (Spain), Bogotá (Colombia), Hyderabad (India), London (United Kingdom), Rio de Janeiro (Brazil), Skopje (Macedonia), New York City (United States of America). We consider and refer to the RAMSES cities (or countries they are situated) within the relevant chapters. The chapters that form this report are briefly introduced below.

Chapter 2 reports on the “International, national policies and cities strategies”, where we assess features and characteristics of cities to determine the influence climate change policies and supporting networks can have on the city strategy. By quantifying policy efforts from across 200 European cities we demonstrate the influence of national climate change policies on local plans to introduce infrastructure systems that help adaptation and mitigation efforts. We provide evidence that a multi-scale approach to climate policy is needed at both the city and national level. We find that there is no archetypical way of planning for climate change, and multiple interests and motivations are inevitable. We suggest that European, national and regional governments need to provide sufficient capacity and resources to enable cities to plan, respond and act to the challenges and opportunities climate change can bring. But how can we benchmark the actual performances of cities?

We provide a brief review and analysis of quantitative methods that can assess efficiencies in terms of environmental impacts and present “Methods and examples of benchmarking cities” in Chapter 3. Addressing issues concerning global sustainability relies heavily on the identification of best practices which can be adapted to other cities. This chapter aims to benchmark and identify cities that can provide better socio-economic opportunities with less pollution by-product. We test the Data Envelopment Analysis (DEA) to rank 88 European cities and find irrespective of the ranking method that strong socio-economic cities are more efficient cities. Thus cities with relatively lower socio-economic benefits are less efficient i.e. more polluting. The city benchmarking study support local governments to find innovative ways to address sustainable development challenges. However it is inevitable that cities would like to know the scale of challenges and get some broad idea on the costs or range of costs for various pieces of infrastructure they may want to provide or promote.

Thus in Chapter 4 we provide “Cost inventories for 9 infrastructure components”. We developed cost inventories for the installation, construction, retrofitting of specific urban infrastructure components that support adaptation efforts to reduce the impact of urban heat waves and flooding. Such components are air conditioning, mechanical ventilation, green roofs, coastal and river embankments (i.e. levees) and permeable paving. We additionally report on costs for infrastructure components that support mitigation efforts, such as double glazing, solar blinds and solar panels. We find that RAMSES cities with higher Gross Domestic Product (GDP) per capita had the largest amount of data available. This suggests that cities with lower GDP face a large bulk of preparation work, but might not have the tools (and finances) available to effectively implement infrastructures that support adaptation and mitigation efforts.

One infrastructure component that can have multiple benefits is Green Urban Infrastructure but little is known how to benchmark its performances and select suitable species. We describe in Chapter 5 the “Assessment and selection methods for Green Urban Infrastructures (GUI)” and present a Performance Index (PI) that is based on a case study. The PI is a quantitative method to select GUI species for city components and can serve as an evaluation framework for streetscape vegetation. We use quantitative measures to assess infrastructure components by focusing on five traits of the GUI that act alongside streetscape vegetation. The traits are Pollution Flux Potential, Carbon Sequestration Potential, Biomass Energy Potential; Environmental Stress Tolerance and Crown Projection Factor. Based on this we quantify and rank the infrastructure components applying the PI to the traits in terms of usefulness of the type of GUI to reduce the risks or increase the efficiencies to climate change adaptation or mitigation. We test our method sand the PI by ranking fifteen street vegetation species. We believe that the PI offers a transparent metric for developing sustainable streetscape green infrastructure that can be used by researchers and city planners alike.
Other components that support mitigation efforts are Electric Vehicles (EV) and their supporting infrastructure. A case study is presented in Chapter 6 where we investigate “Cities strategies and Electric Vehicles”. We evaluate the effectiveness of mitigation strategies of 30 UK cities to encourage the uptake EV and the provision of the needed infrastructure. Our results show that thirteen out of 30 urban audit cities mention EV as part of their climate change mitigation strategy. Yet, there is no statistical difference for the number of charging points or numbers of registered EV for the cities that have EV as part of their climate change mitigation strategy and those that do not. We also show that EV, when compared to an efficient ICE (Internal Combustion Engine), might not be the magic bullet to achieve the needed reductions by 2020 or 2027, and much more incentives are needed to the switch to EV in order to achieve the necessary carbon reduction targets.

To stay with the mitigation theme we explore in Chapter 7 “The trinity of scaling to assess cities” and ask whether smaller or larger cities are more efficient in terms of per capita Carbon Dioxide (CO₂) emissions. In the literature little attention has been given to quantify the implications of accelerated socio-economic activity in cities and in this chapter we concentrate on CO₂ emissions from cities and to what extent urbanisation drives or mitigates climate change. With the aim of providing quantitative methods to analyse the efficiency of city structures with regard to their potential to reduce CO₂, we analyse scaling with population size (as one of the indicators) and find that the exponent is dependent with a transition from super- to sub-linear scaling. We find that urbanisation can be an important factor to drive climate change in developing countries and an important factor to mitigate climate change in developed countries. However further studies, measuring the exponent for countries rather than aggregating cities, are needed to confirm our results.

We then describe other methods to assess cities and their efficiency drivers in Chapter 8. We investigate “City characteristics and CO₂ efficiency” by employing the City Clustering Algorithm (CCA) and utilise gridded CO₂ emissions data for cities in the USA. We analyse all inhabited areas in the US, and show a sub-linear relationship between population density and the total emissions (i.e. the sum of on-road and building emissions) on a per capita basis. We find that doubling the population density would entail a reduction in the total CO₂ emissions in buildings and on-road sectors typically by at least 42%. Moreover, population density exerts a higher influence on on-road emissions than buildings emissions. From an energy consumption point of view, our results suggest that ongoing urban sprawl will lead to an increase in on-road energy consumption in cities and therefore stresses the importance of developing adequate local policy measures to limit urban sprawl. Our study suggest that one can no longer view a city in isolation with clearly defined boundaries but rather as a dynamic cluster of entities constantly evolving in space and time. Therefore urban policies aiming at curbing the energy resources consumed by these clusters should not only aim at smart growth policies within the city but also at a broader metropolitan/regional level including the satellite towns which are well beyond the current city boundaries.

In Chapter 9 we provide our “Main conclusions from this report (D1.3)”. Across the 7 different chapters we benchmarked and analysed the vulnerability, adaptiveness and efficiencies of cities and developed inventories that help assessing city measures, components and strategies. We believe that the qualitative and quantitative research reported here will not only improve the assessments but also prompt the implementation of infrastructure components that support climate change mitigation and adaptation efforts in cities worldwide. The research conducted and reported here has direct relevance to the ongoing and future research of the RAMSES consortium, namely T5.4, T7.3, T8.3, WP 9 and WP 10 as described in the individual chapters and summarised in the concluding chapter.
2 International, national policies and cities strategies

To identify the key elements and drivers for mitigation and adaptation efforts at the city level we proposed in the Description of Work (DoW) of RAMSES (Kropp, 2013) to combine sector-based assessments with regionalized assessments. As described by RAMSES D7.2 (Domingos et al., 2015) climate change adaptation requires a context-oriented approach both in terms of policy prescription and evaluation. However there is a poor understanding of the relationship between the overarching international and national policies and the strategies and plans that are made by cities. By investigating published strategies and plans we analysed if cities are influenced by national and international policies. Similar to RAMSES D8.2 we argue that a systems-based approach needs to consider trade-offs, synergies and interlinkages with social and environmental issues and need to overcome regulatory, structural and operational barriers (Mendizabal et al., 2016).

We assessed the relation of EU and national public policies in order to provide useful insights that advance climate policies at European, national, regional and local levels. This has direct relevance to future RAMSES work such as WP8 (Stimulating European urban strategies for transition), WP9 (Stakeholder dialogues) and WP10 (toolbox). This chapter is based on a research paper that will be published in May 2016 in the peer reviewed journal “Journal of Environmental Management” (Heidrich et al., 2016). In this chapter we provide a benchmark, interpretation and illustration of the influences from international and national networks and policy makers that can stimulate local strategies or implementation plans. This will consequently encourage the provision of infrastructure systems that help mitigating or adapting to climate change in cities.

2.1 Introduction to climate change policies, strategies and plans

Dealing with climate change is one of the many challenges for the European Union (EU), which has set ambitious short and long-term emissions reduction targets. Cities are crucial actors of climate change mitigation and adaptation efforts (Kousky and Schneider, 2003; Rosenzweig et al., 2010). However, how and why cities engage in climate policy remains largely unclear and the effect of (binding or non-binding) policies from higher levels of government is hardly understood (Kelemen, 2010). As discussed by the RAMSES Stakeholder interviews (D9.1) reasons to adopt adaptation efforts vary among European cities (Terrenzi and Wigström, 2014). And whilst academic scholars note a supporting effect (Biesbroek et al., 2010; Heidrich et al., 2013; De Gregorio Hurtado et al., 2014), the mere existence of international or national climate policies is no guarantee for local plans and action. As part of a wider research network COST TU0902 (Salvia et al., 2013; De Gregorio Hurtado et al., 2014; Olazabal et al., 2014; Reckien et al., 2014a; Reckien et al., 2015; Heidrich et al., 2016) for RAMSES we led the assessment of EU, national policies and city strategies and plans and benchmarked climate change mitigation and adaptation efforts across Europe. In this chapter we address the following questions:

1. How are climate change mitigation and adaptation efforts incorporated into national policies and city strategies or plans, and
2. What are the influences of national climate change policies on local strategies or plans to introduce infrastructure systems that help adaptation and mitigation efforts?

We investigated climate change strategies and plans at city level and conducted a comparative study of cross-national policy from Austria, Belgium, Estonia, Finland, France, Germany, Ireland, Italy, Netherlands, Spain and the United Kingdom (Heidrich et al., 2016).

2.1.1 International climate change agreements, alliances and EU policies

The United Nations Framework Convention on Climate Change (UNFCCC), adopted in 1992 (United Nations, 1992) the Kyoto Protocol to set binding reduction targets but provide important flexibility how national policies achieve these (Albrecht and Arts, 2005; European Commission, 2010a). The EU plays a leading role in global mitigation efforts (Bäckstrand and Elgström, 2013; Rayner and Jordan, 2013) and pushing for internationally binding emission reduction targets (Bäckstrand and Elgström, 2013). The European Parliament (2009) set the 20-20-20 target in which aims by the year 2020 to have reduced
GHG emissions by 20%, increased energy produced from renewables by 20% and increased energy efficiency by 20% (1990 baseline). Moreover, in 2011 the European Commission (2011b) committed to 80% GHG emission reductions by 2050. Achieving such ambitious targets requires major interventions from public and private actors (Harvey et al., 2014 {Guy, 2015 #1555}); across scales and sectors as highlighted by IPPC (2015) and UNFCCC (2015). It is increasingly recognized that successful climate mitigation and adaptation does require Member States to engage urban areas (i.e. cities) in their national climate change policy (European Commission, 2011a; Stecker et al., 2012). The EU supports this view with initiatives such as the Energy Strategy 2020 (European Commission, 2010b), the Covenant of Mayors (2013) and the Mayors Adapt (2014), Strategic Energy Technology (SET) Plan and Smart Cities Initiative (European Commission, 2009b). The numerous climate change networks are a witness of this credo. They support, coordinate and bundle city actions on the ground to introduce infrastructure systems, but also help in gaining financial support and transfer expertise.

2.1.2 National and local climate change policies

Governmental, institutional and legal structures of public policies and its influence on urban plans differ across EU-27 countries. For example, Albrecht and Arts (2005) found that state centric countries, where governments steer and guide society from above, like France and countries in economic transition like Estonia seem to have less precise implementation plans and time frames in tackling climate change (Rayner and Jordan, 2013). Nevertheless cultural, historical and planning traditions should be considered (Plümper and Schneider, 2009; Getimis, 2012) in assessing climate change efforts. Scholars note the positive influence of national frameworks (Stecker et al., 2012) and climate networks for climate change mitigation, the latter particularly in larger cities (Rosenzweig et al., 2010; Cerutti et al., 2013). However, not all cities develop climate policies in countries that provide national climate policies, or that support international targets or joined supporting networks (Sharp et al., 2011; De Gregorio Hurtado et al., 2014).

Mitigation often seems more advanced than adaptation as it can complement or integrate sector specific policies for example transport or waste management or even master plans. Adaptation and its integration in European Member States’ planning systems is still a challenge, though many national strategies identify policies and planning instruments pivotal to its integration (Biesbroek et al., 2009; Reckien et al., 2014a). The recently published European Adaptation Strategy (European Commission, 2013) and the initiative, Mayors Adapt (2014), is engaging cities more on climate change adaptation (Domingos et al., 2015). Overall, integration of climate change into policies and planning processes remains a challenge (Heidrich et al., 2013; Zanon and Verones, 2013). However, Rayner and Jordan (2013) appreciated that the experiences gained at EU level can be of great importance to those seeking to understand public policy and governance of climate change, both within and between countries. It is therefore important to understand the implications of national frameworks to climate change policies, planning and performance by European cities (Albrecht and Arts, 2005).

2.2 Investigating climate change strategies across 200 EU cities

To investigate the relation of climate change strategies at city level and EU and national public policies, we gathered a sample of 200 cities (defined by administrative and/or political boundaries) from 11 countries (Austria, Belgium, Estonia, Finland, France, Germany, Ireland, Italy, Netherlands, Spain and United Kingdom) from the Urban Audit (UA) database1 by considering the origin and research

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1 The UA database provides European urban statistics for 258 cities across 27 European countries containing almost 300 indicators presenting information on matters such as demography, society, the economy, the environment, transport, the information society and leisure. UA database cities are selected, in close collaboration
experiences of our colleagues across the RAMSES consortium and beyond (Heidrich et al., 2016). The 11 countries include 72.1% of the EU-27 population and the 200 cities represent 16.8% of all EU-27 inhabitants (2008). We compiled a database of international and national climate change mitigation and adaptation policies and the published strategies and plans from cities. Data, national and local climate change mitigation and adaptation strategies were translated and interpreted by native speaking authors and the latest entry being made in January 2015. We focused our analysis on the following elements:

1. Emission reduction targets (in terms of CO₂ equivalent);
2. Level of achievement of such targets;
3. Sharing of responsibility for climate change policy across administrative tiers; and
4. Membership to networks, such as the CoM- Covenant of Mayors (2013).

We investigated urban climate change mitigation and adaptation plans and policies that were officially adopted, published or in development (if a draft was made available). The documents were obtained by either contacting city representatives or policy makers directly or by retrieving published information from the cities websites². Information on membership in the international climate network CoM was retrieved from its website.

2.3 Results of national and local policies on city strategies

2.3.1 National climate policies of 11 EU countries and examples

The 15 countries that were part of the European Union when the Kyoto Protocol was adopted in 1997 (the ‘EU-15’) committed to reduce their collective emissions of six GHGs in the Protocol’s first period (2008-2012) to 8% below 1990 levels. This target was then translated into specific targets (Table 1, column 2) for EU Member States (European Commission, 2010a; UNFCCC, 2012), which are characterized by different national public policies, governance systems and institutional settings. We illustrate below how national public policies are being translated to the local (i.e. city) level by reporting on individual climate change plans and/or strategies published from the 200 cities in 11 countries that we surveyed (Figure 2-1).

Ten out of the 11 countries have plans to reduce emissions 10 - 20% between 2005 and 2020 and only Estonia is allowed to increase emissions during that time, because of a development concession—although their emissions actually reduced. At the end of 2011, only seven countries out of the 10 investigated reduced their emissions; Austria, Ireland and Spain, increased their emissions. Belgium, Finland, France, Germany, and the UK either met or over-achieved their targets. Italy and the Netherlands were able to reduce their emissions, but insufficiently to meet their targets. Here we select some countries and refer to the research paper which describes all the countries and their efforts (Heidrich et al., 2016). In Austria, where national emission reduction targets were not met, and Germany, where emission targets were surpassed, responsibilities are shared between the federal states³ (i.e. Länder) and cities. The focus of climate planning in Austria appears to be at the national level with its Climate Protection Act (Klimaschutzgesetz) rather than at a city level, although a number of its ‘Länder’ (Vienna, Upper Austria, Lower Austria, Salzburg) also have their own regional climate change

between the Directorate-General for Regional Policy, Eurostat and the national statistical institutes (Eurostat 2011), according to the following rules set by (Eurostat 2007): (i) coverage of approximately 20% of the national population; (ii) inclusion of all capital cities and, where possible, also of regional capitals; (iii) inclusion of both large (more than 250,000 inhabitants) and medium-sized (between 50,000 and 250,000 inhabitants) cities and geographical dispersion within countries.

² National, regional and city administrations (and often different departments of the latter) do publish climate change policies, plans and strategies continuously with time lags between authorization, publication, etc., making data accumulation difficult. Although greatest effort was made to accumulate all urban strategies available, the database and results may not be complete or up-to-date (as cities have published new plans since).

³ The Federal State is a political entity characterized by a union of partially self-governing states or regions under a central (federal) government.
programmes (Kriech et al., 2009). In 2002, Linz set an ambitious target of 50% reduction by the year 2030, whilst Vienna and Graz have set CO$_2$e reduction targets of 21% and 30% by 2020 (with a 1990 baseline). It is interesting to note that there is not one Austrian city that published an adaptation plan.

In Germany, the federal state shares responsibility for climate change policy with the Federal States (Lander). However, it has been more successful in meeting its target than Austria, achieving a reduction of CO$_2$e of 24% between 1990 and 2011 (UNFCCC, 2013). Germany has a national climate protection strategy and targets for emissions reductions, which was adopted in 1990 by the Federal Environment Ministry (BMU) and a national climate change adaptation strategy (Bundeskabinett, 2008). Approximately 80% of GHG emissions in Germany relate to energy and a large proportion of its reduction has been attributed to the reunification of Germany and “clean up” of coal power stations and economic change in the former East Germany (Darwall, 2013). We found that 80% of the 40 cities analysed provide a mitigation plan with qualified GHG reduction targets, but only 32.5 % have an adaptation plan. Only 12 cities have both an adaptation and a mitigation plan, like the capital city of Berlin as well as some larger cities like Stuttgart in the South, Hamburg in the North, Dresden in the East and Düsseldorf in the West. The most ambitious cities are Berlin and Hamburg both targeting a GHG emissions reduction of 40% in 2020 and 85% and 80%, respectively in 2050.
In Belgium, the National Commission introduced the first National Climate Plan in 2009 (Commission Nationale Climat, 2009) and a National Adaptation Strategy in 2010 (Hoyaux et al., 2010). Similar to Austria, the focus of climate planning is centred at the regional and federal government level. Among the seven Belgium UA cities, we could only identify a few initiatives in the Brussels region and in a number of cities in Flanders. Brussels’ mitigation plan was published in 2002 and (only) refers to a 7.5% CO2e reduction target to be achieved in 2010 (1990 baseline). Consecutive to joining the Covenant of Mayors (CoM) initiative, Brussels published another mitigation plan in March 2010 describing measures that are adopted to reach the 20% emissions reduction target of the EU by 2020 and to decrease this further to 30% by 2025 (1990 baseline). However, Brussels does not depict the situation in Belgium as it has not only city, but also a regional status. Some examples such as Antwerp and Gent show that a few cities are nonetheless active in Flanders, the former even aiming at being carbon neutral by 2050.

Estonia has reduced its CO2e by 47% (Table 1, column 3), but this is mainly due to a decrease in energy exports, whilst energy production still accounting for 89% of the total GHG emissions in 2011 (UNFCCC, 2013). The government is centrally responsible for climate planning and coordinates regional and local actions. However, some local governments instigated their own adaptation plans to respond to floods and storms since an extreme storm in January 2005. But the completion of local plans is a long process (mainly due to limited capacity and low climate risks). From the two Estonian UA cities analysed (Tallinn and Tartu) it is only the capital city (Tallinn) that has a mitigation plan; none of them has an adaptation plan.

The legal framework in the United Kingdom (UK) is similar to that in Ireland, in part due to similar governance structures and historical ties. Legislation in the UK- the Climate Change Act (AoP, 2008)- requests to establish climate adaptation and mitigation plans. The responsibility for climate change is divided between the national governments and its agencies of Northern Ireland, Wales and Scotland. Locally climate change plans are provided by cities, and London, Leicester and Manchester (who signed the Nottingham Declaration\(^4\)) demonstrate a high level of integration of adaptation and mitigation within their planning processes. Also the three Scottish cities (Glasgow, Aberdeen and Edinburgh) translated public policy into their strategies and all three signed the Scotland’s Climate Change Declaration. Although most UK cities recognised that adaptation and mitigation is related, the larger emphasis tends to be placed upon mitigation. The UK achieved a 28% reduction of CO2e between 1990 to 2011 (UNFCCC, 2013). However, similar to Germany, this is mainly attributable to political and economic circumstances, e.g. recession in the early 1990’s and the large-scale switch from coal to gas fire plants (Darwall, 2013). All cities acknowledge climate change being a threat (Heidrich et al., 2013), although there is large variation in the detail of analysis, targets and timeframes as well as mitigation and adaptation measures under consideration and the degree of implementation across the country.

In Italy the Inter-Ministerial Committee for Economic Planning (CIPE, 2002) approved the National Climate Change Strategy in 2002 and established a cross-ministerial body responsible for monitoring and evaluating the policies in the national strategy (Hogan et al., 2012). The Ministry for the Environment, Land and Sea (IMELS) administered the implementation process of a National Adaptation Strategy (NAS) in 2015. Similarly to France, Ireland, Estonia and Finland, guidance and national plans are provided by the national government and its agencies, which direct regional and local climate plans. No specific examples could be found that illustrate comprehensive mitigation plans, though most cities are active on mitigation issues, mainly through Municipal Energy Plans and, in recent years, Sustainable Energy Action Plans (SEAPs) as members of the CoM (e.g. Genova, Bologna, Modena and Bari). There is little climate change adaptation, which in part is due to the lack of guidelines given at the national or regional level. Adaptation initiatives are often carried out at a higher administrative level of Provinces or Regions e.g. the Province of Genoa- see also Annex of RAMSES D5.1 (Floater et al., 2014).

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\(^{4}\) The Nottingham Declaration is an initiative in England to tackle climate change and its signatories acknowledge for example, the risks of climate change, work to reduce emissions, monitor progress and publish results.

- 14 -
The Netherlands (NL) has like many other countries a three-tiered spatial planning system, i.e. a central, regional (provinces), and local level (municipalities). While the regions are mainly in charge for balancing urban and green space development, the municipalities have a relative freedom to decide about the urbanization policies and development plans (similar to Germany). Therefore, the municipalities are the main actors in the Dutch climate change policy development. The cities are very active and ambitious, e.g. many aim to become carbon-neutral within 40 years (Reckien et al., 2014b). On the contrary climate change adaptation is hardly practiced by Dutch municipalities. The city of Rotterdam is the only Dutch city that has an explicit and detailed climate change adaptation plan. The main reason for the lack of urban adaptation programmes in the NL is the so-called Delta Programme. This is a national programme that handles strategic adaptation planning and implementation to address rising sea levels and other water supply and water quality issues. Based on the awareness of these national instruments, Dutch cities may perceive no need to provide local plans. In addition, Biesbroek et al. (2011) identified conflicting timescales and interests, lack of financial resources, unclear division of tasks, uncertain societal costs and future benefits, and a fragmentation within and between scales of governance in the Netherlands as the main barriers for the development and implementation of climate change adaptation strategies.

### 2.3.2 Climate change plans from the cities surveyed

From the 200 cities surveyed, 130 (64% of our sample) have climate change mitigation and/or adaptation plans (Table 2-1). The United Kingdom (UK) clearly dominates the sample with over 90% of the cities having a mitigation plan. The Netherlands and Germany follow second, where 80% of the cities have a mitigation plan. Belgian cities are not (yet) very active in the development of climate mitigation plans, and are being less effective in translating national policy into local action. The ratio of French cities with plans should have increased as deadlines in the National law (LOI, 2010) demanding cities to provide plans have passed since this research has been completed.

<table>
<thead>
<tr>
<th>Country</th>
<th>Cities</th>
<th>Mitigation</th>
<th>Adaptation</th>
<th>Mitigation CoM</th>
<th>Adaptation CoM</th>
</tr>
</thead>
<tbody>
<tr>
<td>Austria</td>
<td>5 -16</td>
<td>+16.3 2001</td>
<td>60.0 2012</td>
<td>0.0 0</td>
<td>14</td>
</tr>
<tr>
<td>Belgium</td>
<td>7 -15</td>
<td>-16.4 2008</td>
<td>42.3 2010</td>
<td>0.0 4</td>
<td>65</td>
</tr>
<tr>
<td>Estonia</td>
<td>2 +11</td>
<td>-47.3 2004</td>
<td>50.0 -</td>
<td>0.0 1</td>
<td>3</td>
</tr>
<tr>
<td>Finland</td>
<td>4 -16</td>
<td>-23.2 2008</td>
<td>75.0 2005</td>
<td>50.0 4</td>
<td>7</td>
</tr>
<tr>
<td>France</td>
<td>35 -14</td>
<td>-16.7 2011</td>
<td>60.0 2011</td>
<td>35.3 18</td>
<td>151</td>
</tr>
</tbody>
</table>

Overall, only 23% of cities (56 out of 200) of the sample have an adaptation plan (column 7, Table 1), though this differs substantially between countries. The UK is the most active country, 80% of the cities (24 cities) have an adaptation plan. In Finland and Germany 50% and 33% respectively have an adaptation plan. The fewer number of cities with adaptation plan could lead to the assumption that

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5 [http://www deltacommissaris.nl](http://www.deltacommissaris.nl),

6 GHG limits in 2020 compared to 2005 as agreed by Decision No 406/2009/EC on the effort of Member States to reduce their greenhouse gas emissions to meet the Community’s greenhouse gas emission reduction commitments up to 2020.

7 GHGs including LULUCF, in Gg CO2e Change from base year (1990) to latest reported year (2011) in % as described in UNFCCC (2013) ‘Total CO2 Equivalent Emissions with Land Use, Land-Use Change and Forestry’ United Nations Framework Convention on Climate Change (UNFCCC) Time series - Annex I. Bonn, Germany: UNFCCC.
mitigation plans are a precursor for adaptation plans. This assumption holds true for many local plans, but only partly for national policies (Columns 4 and 6, Table 2-1) as mitigation policies are more updated. Where there is a long history of mitigation activity, e.g. in Germany, adaptation plans are more likely.

Many cities review and update their mitigation plans but this is rarely the case for adaptation plans. A large proportion of cities take part in global and European climate networks (e.g. Eurocities, Energy Cities, Climate Alliance and CoM). Participating in these transnational networks and alliances supports the development of an explicit urban approach to climate change, providing cities with easy access to best practices and helping them to develop local capacity (Cerutti et al., 2013). We analysed the influence of the CoM in more detail, as a wide range of cities across Europe were part of this initiative. Signatories of the CoM go beyond EU 2020 targets committing to reduce CO₂ emissions by at least 20% and promoting measures and projects aimed at increasing energy efficiency and the use of renewable energy sources. Table 2-1 illustrates that countries with a high number of mitigation plans i.e. Germany, Netherlands, and UK, have relatively few cities that are members of CoM. On the other hand, Italy and Spain are the countries with the largest number of signatories of the CoM (respectively 2582 and 1323 municipalities), together representing more than 77% of the overall number of COM signatories (Cerutti et al., 2013; Reckien et al., 2014b). Similar to the RAMSES cities (Domingos et al., 2015) our analysis highlights that European cities have large differences in terms of their degree of advance in climate change policy, and the different motivations that lie behind the development of mitigation and adaptation plans. It seems that adaptation, although delayed, runs behind mitigation strategies (e.g. UK and Germany) and that, while national legislation might be instrumental for the development of local climate plans (e.g. France), international networks and activities are able to act as a motivator in the approval of plans (e.g. Italy).

2.3.3 Influences of national climate change policies on local climate strategies and plans

Overall, the influence of national policy frameworks and guidelines is inconclusive, as causal relationships cannot be established. Whether a country is a late-comer or a forerunner in the development of national climate change policies had varying influences on the likelihood of developing local adaptation and mitigation policies. For example, Germany’s first national mitigation strategy (2000) is rather old compared to the UK’s Climate Change Act (AoP, 2008), but both countries rank high in mitigation plans. The same holds true for adaptation. While Finland’s adaptation strategy is the oldest in the sample (2005), only 50% of Finnish cities have adaptation plans, while in contrast 80% of UK cities have adaptation plans. Countries that acted rather late in terms of national climate change policies have been considered more ‘fence sitters’ rather than ‘pace setters’ in European environmental policy making (Börzel, 2002; ESPON Climate, 2011), i.e. they have in the past followed EU policy rather than leading it in terms of influencing the agenda or implementing policies. Yet, Spain has been relatively late in adopting policies but has now developed and implemented a number of mitigation policies (ESPON Climate, 2011). Countries that acted late in formulating a comprehensive national mitigation strategy have seen some of their cities develop disconnected climate change policies and measures. Others, e.g. Ireland and France try to “catch up” quickly and cities release many sectoral policies and set ambitious long-term targets but again there is limited cohesion in achieving the country’s long-term targets.

France provides an interesting case where national policies make the development of local climate change action plans obligatory. Before the law made their development compulsory, only sectoral actions as parts of local Agenda 21 were undertaken by the cities (with the sole exception of the capital Paris that released a comprehensive independent plan in 2007). The 2010 Grenelle II law (LOI, 2010) led to the adoption of a flourishing number of local climate plans between 2010 and 2013. In Spain, there is no legal requirement for cities to develop mitigation or adaptation plans. However, the central government is providing a framework for climate mitigation and adaptation and a number of institutional initiatives support the involvement of the regions in order to achieve the national goals. Regional advances are also supported by EU initiatives, for example in Italy, but there is no central obligation regarding the implementation of local climate plans. In most of the cases the proactive role of cities in
implementing climate plans depends on their involvement in international associations (e.g. CoM, Local Agenda 21), and/or in European projects. For example, the adaptation plan of Padova and more recently the one of Ancona have been developed following the cities’ involvement in specific EU projects.

2.4 Implications and conclusions of investigating policies and strategies

The analysis of 200 cities and their mitigation and adaptation strategies across 11 European countries showed a substantial commitment by European cities. However it is difficult to establish the influence of European and national policies on urban climate change adaptation and mitigation planning and the associated infrastructure systems. This is at least in part due to scale, as mitigation and adaptation are cross-sectoral (ESPON Climate, 2011). Policy integration often takes place at higher regional and national level and not at the local level. Regions might often be the more relevant scale for providing infrastructures, tackling and managing climate change issues, e.g. for urban sprawl and its relationship to mitigation, or for the catchment scale and corresponding flood risk management with respect to adaptation (Biesbroek et al., 2009). In our analysis the UK sticks out in many respects, i.e. the number of adaptation and mitigation plans, sectoral policies, nationally agreed long-term targets, and the planning system is based on larger spatial units than compared to other EU member states, such as Ireland, France, Germany, the Netherlands, and Spain (Oxley et al., 2009), which probably makes adaptation and mitigation planning more effective and coherent. The relevance of the size of spatial planning units for the suitability or efficiency of mitigation and adaptation plans might also contribute to smaller cities having less often climate change plans. Some climate change issues are better managed at broader scales.

Figure 2-2 Global, EU and national policies/networks and city climate change plans
Our findings highlight the shared responsibility that global, European, national, and regional policies and international networks have in stimulating the development of local climate plans as illustrated in Figure 2-2. The figure shows that global commitments or EU policies may result in the creation of international networks similar as EU commitments have resulted in EU networks of cities. Within the countries we investigated there is clear evidence that international policies, national law, and networks, and sometimes even regional networks, do support cities in developing their strategies and plans.

As discussed, the existence of global policies or national legislation and similar governance structures do not guarantee the development of city plans. Furthermore, although an influence of national government frameworks on city plans was documented in some countries (Stecker et al., 2012), a national framework is not always sufficient to trigger climate change action on the ground (De Gregorio Hurtado et al., 2014), which is supported by our study. For instance, the French experience of the Grenelle policy showed how strong national legislation can lead to a thriving number of comprehensive local climate plans within only a few years. Although many French cities experience delays in the application of the national policy, mostly due to lack of local capacity and long consultative processes, detailed national guidance and support by the French environmental agency (ADEME) makes it easier for cities to develop plans. Also, the UK has a high number of urban adaptation and mitigation plans, high sectoral policies and a nationally agreed long-term target, indicating that national policies are effective. In turn, the example of Italy shows the contrary: a national framework might not be required to stimulate climate change plans development.

Overlaps of national, regional and city climate policies can exist as central government policies influence the selection of mitigation and adaptation measures within cities (Bulkeley and Kern, 2006; Biesbroek et al., 2009; Bulkeley, 2009). However, we also provided evidence that the lack of national leadership can lead to a more active membership in climate change networks. The CoM initiative is particularly popular in the Southern European countries where a national strategy is missing, such as Italy, as well as in countries that implemented national mitigation strategies relatively late, e.g. Spain (Reckien et al., 2014b). Moreover, cities in countries with no nationally agreed long-term targets often align GHG reduction levels at the EU20-20-20 targets, as seen in Spain, Italy and Estonia. This suggests that European climate policies have a large influence in countries without, or with weaker, national policies.

Many other strategies such as transport, waste, energy policy can have significant impacts and relevance to climate change. Those strategies can be in areas such as: reduction of emissions from transport (e.g. transport plans); protection from hydrologic risks (e.g. flooding, drought) and hydro-geological risks (e.g. landslides, aquifer vulnerability); increasing carbon absorption (e.g. green urban areas); creation of fresh air corridors (e.g. urban parks); support and assistance during the summer months for disabled and elderly people (e.g. heat wave plans, energy efficiency or warning systems). However, our study collected, investigated and analysed strategies and planning documents published by the cities under the banner of ‘climate change’, ‘mitigation’ or ‘adaptation’. This limits the conclusions of this study, as action in other sectors or bigger scales (e.g. transport, national rail infrastructure, coastal floods) might have been developed, but were not included. For example, in Italy and Netherlands, flood protection responsibilities pertain to regional and national levels and therefore, no such action or measures are implemented at local level.

Our analysis shows that many European cities are proactive on climate change. One reason is that cities are the closest to citizens, which can positively influence emission reduction efforts at the local level and safeguard and manage the risks of extreme events and disasters as local perception on the potential costs and benefits of mitigation and adaptation action increases. Beyond the influence of upper levels of government and networks of cities, the local level seems to be acting on the basis of the experience gained implementing environmental policies over the years. This experience can provide cities with the skills necessary to cope with climate issues, even in countries that have not established an explicit link between national and local climate action. In those cases, the experiences gained by cities on environmental matters have made them aware of the importance of being active in the climate change policy field (De Gregorio Hurtado et al., 2014). However climate change mitigation and adaptation often lies outside the administrative boundary of the city and clear guidance and collaboration across the city boundary is needed. Cities look for national guidance but if this is not available, the most
proactive ones align themselves to international guidance and networks such as ICLEI (2008) and CoM. However, tackling global issues requires more than the planning and action from the most forward-looking cities. Stronger and coherent national and where applicable regional strategies are required. Our analysis confirms that multi-scale provision of plans and strategies (Bulkeley, 2009) from European, national to regional level is most effective in ensuring that cities will plan for mitigation and adaptation. Cities can provide and deliver plans without the wider support and guidance but they need to have the capacity, resource and political will to do so. Our analysis suggests that, where such wider support is limited, larger or capital cities achieve this. This can create a disconnect between smaller cities and larger cities, which public policy should fill by providing clear climate change strategies for cities of any size. One potential solution seems the employment of larger planning units for climate change mitigation and adaptation issues, e.g. the organisation of collective action of a number of smaller cities.

We confirm that adaptation plans and implementation strategies are unique and their effectiveness tends to depends on participatory and inclusive measures, the recognition both of local and scientific knowledge, accountability, and stakeholders engagement (Domingos et al., 2015). In this chapter we did not discuss the obstacles cities my face as this was outside the scope of this study and we refer to the research presented by RAMSES D7.2 (Domingos et al., 2015); D9.1 (Terenzi and Wigström, 2014) and D8.1 (Mendizabal et al., 2016). Nevertheless our findings of the influence of national policies on city strategies and plans did and continues to inform benchmarking and efficiency assessment research of T1.3 described in Chapters 3, 7 and 8. Our findings also have wider relevance to the current and future RAMSES research, specifically RAMSES Tasks 5.4 and 7.2 as well as Work Packages 8, 9 and 11 can benefit from our findings as city representatives, urban planners and EU policy makers find qualitative and quantitative evidence on the influences of policies on city strategies and plans. Finally we conclude that, there is no archetypical way of planning for climate change and providing the associated infrastructures, and multiple interests and motivations are inevitable. Reinforcing the findings from the stakeholder interviews reported in D9.1 (Terenzi and Wigström, 2014) our research supports the need for a multi-scale approach to climate policy making that ensures the engagement across scales, and that provides sufficient capacity and resource to enable cities to plan and implement infrastructure systems that reduce and/or respond to climate change.
3 Methods and examples of benchmarking cities

Whilst developing relevant policies to address climate change described in the previous chapter; local policy makers often seek best practices/measures that have been implemented elsewhere. RAMSES D1.2 developed a library of impact functions for a typical damage for an event of given magnitude for various cities (Boettle et al., 2016) and RAMSES D3.1 conducted a high level risk analysis comparing 571 EU cities (Tapia et al., 2015) and rankings of cities were conducted. However, defining ‘best performance’ is often constrained by the method that is used to benchmark. As described by RAMSES D8.2 a system-based approach needs to consider trade-offs, synergies and interlinkages with social and environmental issues needing to overcome regulatory, structural and operational barriers (Mendizabal et al., 2016) and although some of the barriers are described in Chapter 2, what are the trade-offs, what is the best practice and how can cities benchmark their performance?

As suggested by the DoW (Kropp, 2013) we summarise some of the state-of-the-art methods that help benchmarking the performances of cities in this chapter. This chapter, which is a shorter version of a paper that will be submitted in May 2016 to a peer-reviewed Journal (Gudipudi and Rybski, Draft) compares and validates results we have obtained for 88 cities using various benchmarking methods e.g. Ordinary Least Squared (OLS), Stochastic Frontier (SFA) and Data Envelopment Analysis (DEA). Although our analysis is limited to benchmarking the socio-economic efficiency of the European cities considering environmental impacts, our method has wider applications in benchmarking climate change mitigation and adaptation efforts at a various spatial and temporal scales. As is the aim of the RAMSES project (Kropp, 2013), it is expected that some of the methods (and the DEA specifically) described in this chapter can be applied to assess mitigation and adaptation options, which is relevant to outstanding RAMSES research in WP8 and 10. The DEA is applicable both, at local scale (e.g. addressing specific measures to e.g. protect sea level rise and the subsequent adaptation costs) and regional/national scale e.g. improving energy efficiency in buildings or transportation.

3.1 Introduction to benchmarking cities

The contemporary science of cities has bid adieu to the previous notion of conceiving cities as organized top-down complex entities constantly seeking for equilibrium by embracing them more as organisms resulting from numerous bottom up evolutionary processes (Batty, 2012). Indeed cities, like biological organisms thrive on natural resources while releasing pollution as a by-product; the interactions of which exhibit allometric scaling (Bettencourt et al., 2007; Samaniego and Moses, 2008). Harbouring more than 50% of the world’s population, generating 80% of the global GDP while consuming approximately 70% of global energy supply (Seto and Dhakal, 2014; UN, 2014) and a bulk of global pollution by-products (land, air and water), cities are recognized as foci for global sustainability and climate change mitigation. In their relentless quest for socio-economic development, cities like biological organisms often compete with each other for financial and natural resources. Urbanization as a global phenomenon is posing complex challenges to local governments who constantly seek for best practices to address them. Therefore, addressing issues concerning global sustainability relies heavily on the identification of best practices which can be adapted to other cities, a procedure usually referred to as benchmarking.

In operations research, benchmarking is defined as a technique characterized by the systematic search for efficient procedures and better solutions for complicated problems and processes (Global Benchmarking Network, 2015). Efficiency of a single entity can be most straightforwardly defined as the ratio of its outputs to its inputs (Boussofiane et al., 1991). Benchmarking can be broadly defined as the process of identifying the entity with best efficiency amongst other such comparable entities and the factors contributing to such performance (Boussofiane et al., 1991). The concept of benchmarking therefore has wider application which is often subject to the parameters used and the research objective. Despite this broader application, defining city efficiency as the ratio of its socio-economic outputs to its environmental inputs, this research aims to address a strikingly simple question: Which cities provide better socio-economic opportunities with less pollution or by-products? With an objective to identify the best practices that can be adapted to local circumstances such a city benchmarking study enables local governments to find innovative ways to address their current sustainable development challenges.
3.2 Contemporary methods for city benchmarking

The process of quantifying efficiency in cities could start from a very rudimentary approach of comparing the ratio measures. For instance, if the per capita total energy consumption of a city is less than that of another city; then the former city is considered to be relatively more energy efficient. However, there might exist many complicated factors that are embedded in urban systems which limits the usage of simple ratio measures. Also, since cities are proved to exhibit scaling laws, such ratio measures might be a mere manifestation of their size or population density and might not reflect ground reality. Therefore, the process of city benchmarking demands more complicated methods to incorporate such city specific heterogeneities. The most common methods used to benchmark city efficiency can be broadly divided into parametric and non-parametric approaches as described below.

3.2.1 Parametric approaches

The parametric approaches that are commonly used in city benchmarking are: a) Ranking based on ordinary least squared (OLS) regression residuals and (b) Ranking based on Stochastic Frontier Analysis (SFA). Having their foundations in econometric theory, the parametric methods for benchmarking calculate the efficiency in cities based on the assumed functional relationship between the given inputs and outputs. Let us assume that all the relationship between the outputs ($y$) and inputs ($x$) considered to identify city efficiency follow a functional form (where $\beta$ is a vector of parameters to be estimated) such as:

$$y = f(x; \beta)$$

Equation 3-1 City efficiency

The inefficiency in the regression approach is determined by the regression residual obtained following the OLS method. Therefore, if a given city ($C_1$) has more residual than another city ($C_2$) then it is considered to be relatively more efficient (or inefficient depending on the defined inputs and outputs). Therefore in case of regression approach Equation 3-2 is written as:

$$y = f(x; \beta) + V$$

Equation 3-2 City inefficiency

where $V$ (regression residual) is the efficiency (inefficiency) component of a city compared to other cities in the regression approach.

However, deviations from the assumed functional form cannot be always attributed to inefficiency component. Statistical noises can sometimes make an inefficient city look efficient in the regression approach and vice-versa. Therefore, the SFA approach (though similar to regression approach till calculation of the residuals) further decomposes the inefficiency component ($V$) into statistical error component ($v$) and inefficiency component ($e$) which results in Equation 3-3.

$$y = f(x; \beta) + v - e$$

Equation 3-3 City inefficiency using SFA approach

where $v$ is the statistical error and $e$ is the inefficiency component. When $e = 0$, the city is 100% efficient and when $e > 0$ there is some inefficiency. SFA assumes that $v$ and $e$ are independent to each other. The model further assumes stochastic relationship between the inputs and outputs and estimates the frontier function and parameters based on maximum likelihood method. Since statistical noise can go in both directions, the model assumes that that statistical error ($v$) is normally distributed [-$\infty$, +$\infty$]. Since, there cannot be a city more efficient than the one in the frontier (the most efficient city); the model assumes that $e$ is half normally distributed (a truncated normal distribution where the point of truncation is zero [0, $\infty$]) in all the cities. For further details please refer to (Bogetoft et al., 2012).
Both OLS and SFA can be calculated only for one output \( (y) \). Therefore, if there is more than one output that needs to be considered while calculating city efficiency, we need to aggregate the residuals obtained using each of the outputs after some kind of normalization procedures (which depend on the distribution of the residuals). If one assumes that all the outputs considered are equally important, one possible way to aggregate the independent residuals post normalization is to simply take the mean of the normalized values.

### 3.2.2 Non-parametric approaches: Data Envelopment Analysis (DEA)

DEA as a non-parametric approach that defines city efficiency as the ratio of its weighted outputs to its weighted inputs. The issue of individual weights assigned to each of the inputs (and outputs) is solved as a linear programming problem. Therefore this approach is considered to be non-parametric in nature. DEA method needs some prior assumptions with respect to the scale of returns (constant or variable) between the given inputs and outputs and the orientation of the efficiency measure. (Bettencourt et al., 2007; Bettencourt et al., 2010) has demonstrated that cities show variable returns to scales depending on the parameters tested (super linear for socio-economic parameters and sub-linear for infrastructure parameters). Thus we can assume that the input and outputs used for benchmarking cities also demonstrate variable returns to scale. Since the efficiency in a city can be increased either by increasing its outputs or decreasing its inputs, the choice of orientation (input or output) depends on the production process characterizing the cities. While in an input oriented DEA, the linear program determines how much input a city can decrease in order to achieve its current level of output; the linear program in an output oriented DEA determines how much output a city can increase (potential output) given its current level of input.

The variable returns to scale (VRS) assumption (Bunker et al., 1984) in DEA identifies the cities in the frontier by piece wise construction of a convex hull (the DEA frontier). The convex hull is defined as smallest polygon that encompasses all the points spread in space. Let \( X \) and \( Y \) represent the inputs and outputs in the data respectively. The subscripts \( m \) and \( n \) represent the \( m \)th input and \( n \)th output of a particular city where \( x_m \) represents the \( m \)th input of a city and \( y_n \) represents the \( n \)th output of a city. Assuming we have a total number of \( M \) inputs \( N \) outputs, the efficiency of a city \((e)\) in DEA is defined as the ratio of its weighted outputs to its weighted inputs (Equation 3-4).

\[
e = \frac{\sum_{n=1}^{N} v_n y_n}{\sum_{m=1}^{M} u_m x_m}
\]

**Equation 3-4 Weighted efficiencies of a city**

Where \( e \) is the output oriented VRS efficiency, \( u_m \) and \( v_n \) are the weights assigned to the \( m \)th input \((x_m)\) and the \( n \)th output \((y_n)\) respectively. Assuming that there are total \( n \) cities in our dataset; the weights \((\lambda_j)\) are assigned to an individual city \((d)\) with these inputs and outputs by solving the following output oriented linear programming problem (Error! Reference source not found.).

\[
e_i = \max_{\lambda, \phi} e \]

Subject to

\[
\sum_{j=1}^{N} \lambda_j x_{ij} \leq x_{id} \quad i = 1, 2, ..., m
\]

\[
\sum_{j=1}^{N} \lambda_j y_{rj} \geq \phi y_{rd} \quad r = 1, 2, 3, ..., n
\]

\[
\sum_{n=1}^{N} \lambda_j = 1
\]

\[
\lambda_j \geq 0
\]

**Equation 3-5 Output oriented VRS efficiency across cities**
The value $\emptyset$ in an output oriented DEA model implicitly assumes that all inputs are used efficiently at their optimal capacity. Therefore the value $\emptyset$ in an output oriented VRS DEA model is more than 1. The inverse of $\emptyset$ is the efficiency ($e$) of the city ($d$). If $e = 1$, the city is in the DEA frontier and is considered to be 100% efficient. If the inverse of $e$ is less than 1 then there is some inefficiency. Since DEA method can accommodate multiple inputs and outputs there is no need for rank aggregation at this stage as we did for the OLS and SFA method.

3.2.3 Exemplary application to European cities

A major pre-requisite for benchmarking city efficiency is a consistent definition of cities and data collection. The EUROSTAT’s urban audit data base available as a part of the new OECD-EC definition of cities (Dijkstra and Poelman, 2012) enabled us to address this pre-requisite. This database includes data for a wide range of topics encompassing physical, socio-economic and environmental characteristics on an annual basis for 3 different types of spatial units:

1. A city (local administrative unit) where the majority of population reside in an urban centre (contiguous urban cells with population density more than 1500 inhabitants/km²) with more than 50,000 inhabitants;
2. A functional urban area which includes the city and its commuting zone, and
3. A greater city (urban centre exceeding the city boundaries) which stretches well beyond the administrative boundary of the city.

We limited our analysis to the 88 most populated European cities (type 1) in the year (circa) 2011 owing to constraints in data availability. For instance, water consumption data is not available for any of the cities in UK (the value for London city used in this analysis is taken from green city index). In case of London, where the local administrative units are too small (City of London with a reported population of 7375 inhabitants with a varying city definition), the Greater Area of London is considered in the analysis. The year 2011, for which most of the cities have reported their data is always considered as the base year for this analysis. In instances where a certain indicator for the year 2011 is not available, the value for the previous year (or next year) is considered, for more information see Eurostat (2014).

Figure 3-1 Urban Audit indicators used to benchmark European cities

Within the urban audit database, three input indicators and two output indicators are identified in this study. The indicators used as inputs include in the analysis (Figure 3-1):

- Annual average NO$_2$ concentration (in µg/m³),
- Annual Water consumption per capita (in liters per capita),
- Annual solid Waste Generation per capita (in metric tonnes).

The following indicators are used as outputs in our study:

- Employment ratio (in percentage), and
- GDP per capita in power purchasing standard (PPS).
All the aforementioned indicators with an exception of the indicator PPS are obtained from the data available under the section ‘Cities/Local Administrative Units’ spatial units mentioned above. The data on the PPS for these 88 cities is obtained from the spatial unit functional urban area (mentioned above) which includes city with its commuting zones. This provides a better approximation of GDP per capita (in PPS) since at a spatial unit ‘city’ level is often difficult as cities usually also include a commuter belt. Detailed description of the indicators used and their respective methodology can be found in the urban audit glossary and methodological handbook (Eurostat, 2004). Instances where the data on a certain indicator is not available for a certain city, information from official documents either from the city/the region or the national statistical offices is used to fill in the data. Since our objective is to identify which cities provide better socio-economic opportunities given pollution by-products; we choose socio-economic indicators as outputs and environmental indicators as inputs. The reason for including two different outputs despite there is a strong correlation between them is to encourage cities which are relatively poor in terms of PPS (for instance east European cities) but have higher employment ratio.

### 3.3 Benchmarking and ranking of 88 European cities

The methods used for city benchmarking and subsequent ranking in the state-of-the-art research can be broadly divided into four categories: (a) per capita ranking measures (Kennedy et al. 2009; Dhakal 2009; Sovacool & Brown 2010), (b) ranking based on normalized and weighted measures (Jiang and Shen, 2013; Singhal et al., 2013) (c) ranking based on deviations in regression analysis (Matas and Raymond, 1998; Larivièere and Lafrance, 1999; Walton et al., 2005; Bettencourt et al., 2010; Gläser and Kahn, 2010) and (d) ranking based on non-parametric approaches such as Data Envelopment Analysis (DEA) (Charnes et al., 1989; Sueyoshi, 1992; Raab and Lichty, 2002; Munksgaard et al., 2005). We found that the results varied significantly with the approach used. Figure 3-3 shows pairwise comparison of the ranking between these cities in DEA method. The green colour in the grid show that the rank of the city under consideration is more than that of another city, the red colour pixel depict that the rank of the city is less than the other city and yellow pixel indicate that the rank of the city is equal to either its own or to other city.

Figure 3-2 illustrates how the ranking of the cities varied depending on the benchmarking methods. Since benchmarking using DEA method allows multiple cities to be on the convex hull, there are more than one city with an efficiency score (or rank) equal to 1. Cities are sorted according to DEA and if the DEA rank is the same they are not further sorted i.e. arbitrary. The OLS and SFA methods have unique ranking for the cities (unlike DEA method with more than one city with an efficiency score of 1).

![Differences in ranking for selected European cities](image-url)

**Figure 3-2** Compare ranking of OLS, SFA and DEA for European cities
3.4 **Conclusions of benchmarking methods and using DEA**

City benchmarking enables identification of best performing cities and the underlying factors which contribute to such efficiency. However, as shown in this chapter, the identification of such best performing cities relies on the methods used for benchmarking. Each method has its own prospects and constraints. While parametric methods require the decision maker to assume the functional relationship of the input and output variables, DEA as a non-parametric method requires the decision maker to have a priori information regarding the returns to scale assumption (constant or variable). A major constraint in using the parametric methods for city benchmarking is that the decision maker can use only one

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8 The green colour pixel while read from the y axis in the figure show that the rank of the city under consideration is higher than that of another city, the red colour pixel depict that the rank of the city is lower than the other city and yellow pixel show that the rank of the city is equal to either its own or to other city. It can be observed that smaller cities such as Wroclaw, Bonn, Leipzig, Gdansk are as efficient as largely populated cities such as Munich and Hamburg. This phenomenon might be attributed to that fact that smaller cities might have less GDP per capita but they also consume/pollute relatively less natural resources (Air, Waste and Water).
output in a single analysis. In case there are more than one variable, the decision maker has to aggregate the individual rankings from each output variable. This is the major advantage of the DEA method as the method can accommodate multiple input-output variables into a single analysis.

However, the convex hull constructed in the DEA variable returns to scale model allows ties (more than one efficient city). If a decision maker is looking for unique set of city rankings then this method might not be appropriate. Also, ranking in DEA relies completely on the number of cities. While the minimum number of cities to be included in this analysis should be at least more than the multiple of the number of inputs and outputs, the inclusion of one new city might completely change the cities spreading the convex hull and the subsequent ranking of all cities. Our analysis of benchmarking socio-economic outcomes considering the environment pollution inputs for European cities using the aforementioned methods, which revealed that the efficiency of cities is independent of the size of the city. Cities such as Munich, Stockholm, Bratislava, Zürich, Copenhagen and Oslo (Figure 3-2) are found to be efficient irrespective of the ranking method majorly owing to better socio-economic conditions. Cities with relatively lower socio-economic benefits such as Leipzig, Warsaw, Lodz and Gdansk are ranked higher in DEA ranking method because of the relatively lower pollution by products. To conclude, defining efficiency of the city as the ratio of its socioeconomic outputs to its environmental inputs, this study shows how the city efficiency varies with the benchmarking methods that are applied. However mature cities with sound socioeconomic background such as Munich, Stockholm, Bratislava, Hamburg, Oslo and Zürich show robust efficiency irrespective of the benchmarking method. Although the methods used in this chapter are confined to the socio-economic outputs of a city given its environmental/resource consumption inputs, the methods can be applied to a wider array of ongoing and future research topics. For instance, the methods introduced in this section can be further applied in testing alternative transition models in RAMSES WP8 and indeed can feed into the training and toolbox of WP 10 to find out the best adaptation option given a set of adaptation options or to find out the best energy efficiency city and the factors contributing to such performance.
4 Cost inventories for 9 infrastructure components

As highlighted in the RAMSES Description of Work (Kropp, 2013) there is a need to provide an inventory of costs for different city components that support climate change adaptation and mitigation efforts. Such inventories will inform city representatives broadly on the level of investment needed to install specific pieces of infrastructure and infrastructure components. For the purpose of this chapter we refer to infrastructure components that are physical structures that serve a city. These are physical components that enable, sustain, or enhance societal living conditions in terms of climate change adaptation and mitigation e.g. air conditioning, green roofs or levees (embankments). The different national and urban strategies and benchmarking efforts across cities and countries are described in Chapters 2 and 3. As was argued by RAMSES D8.1 (Mendizabal et al., 2016) and D2.1 (Kallaos et al., 2014) the choice of favouring specific strategies and infrastructure components are very often based on the perceived (and actual) direct and indirect costs. RAMSES D5.2 did propose an overall methodology for assessing urban costs of climate change (Costa and Floater, 2015) and D2.2 (Acero et al., 2014) described the building and urban typologies across climate impacts. Also, in this report we investigate the performance and impacts of incentives for selected infrastructure components i.e. Green Urban Infrastructures and Electric Vehicles in Chapters 5 and 6.

However very little is known about the actual installation, construction or restructuring costs of specific infrastructure components. Thus we developed a cost inventory for the construction, retrofitting and installation of specific urban adaptation and mitigation measures (components) from air conditioning, mechanical ventilation, and green roofs to coastal and river embankments (i.e. levees). We focus on 9 infrastructure components that reduce the threats of urban heatwaves and flooding as part of adaptation and reduce emission as part of mitigation efforts. We chose these following the RAMSES D5.1 (Floater et al., 2014), which identified these two as major climate threats in urban areas. The associated research and data analysis of the levees cost data has resulted in a draft paper on regressions and uncertainty estimates for coastal defences (Lenk et al., Draft).

4.1 Introduction to cost inventories

Current and projected climate change is heightening the demand for adaptation and mitigation efforts worldwide. In order for policy makers to allocate appropriate resources, it becomes necessary to evaluate, compare and prioritize different adaptation and mitigation structures and components. One widely used method to achieve this is to use cost-benefit analyses (Floater et al., 2014). It is commonly assumed that early investment in climate change adaptation will be likely to be more cost-effective and bring greater incentives in the long run, compared to a responsive approach. However, as is argued by (Linnenluecke et al., 2015), in contrast to climate change mitigation, there are no established frameworks for evaluating the effectiveness of different adaptation options over time.

Across countries and cities only limited evidence of the actual costs of installing the various infrastructure components is available, which indicates a gap between global adaptation needs and the funds available for adaptation. Although a range of studies have calculated the cost benefit ratio to protect for example megacities (Aerts et al., 2014) or the value of providing coastal defence systems (Jonkman et al., 2013) the recent IPCC report (IPCC, 2014a) states that current studies that estimate the cost of adaptation are characterised by shortcomings in data, methods, and coverage. As in the RAMSES Description of Work (Kropp, 2013), this report argues that there is a need for a better assessment of global adaptation costs, funding, and investment (IPCC, 2014a). And while mitigation costs are available at a coarser scale, the barriers for implementing for example energy efficiency are largely due to the initial investment needed and a general lack of cost information (IPCC, 2014b). Although it is clear that mitigation efforts generate costs and trade-offs, they also offer synergies. Specific policies and infrastructure components that support for example GHGs reduction can address other goals, such as managing air pollution, water scarcity, or energy security (IPCC, 2014a).
In the recent review of existing estimates of costs and benefits of adaptation in cities by the RAMSES report D5.1 (Floater et al., 2014) it was found that there is almost no published evidence for the costs of infrastructure components that support climate change adaptation and mitigation efforts across the RAMSES case study cities. One exception is New York City where, following Hurricane Sandy, a range of adaptation costs were estimated, mostly focusing on coastal flooding prevention (Aerts et al., 2013b). Very few cost inventories for specific pieces of infrastructure, rather than whole schemes, that support adaptation and mitigation efforts across cities or indeed countries seem to be available, and where available they are often inconsistent, incomplete, based on different assumptions and years and report in different currencies. And although it is generally accepted that there may be significant differences in terms of the implied distribution of costs across sectors, regions, and over time (IPCC, 2014b) we report in this chapter the costs that we were able to collect for 9 different infrastructure components that support the urban climate change adaptation and mitigation efforts.

4.2 Gathering and standardising cost data for 9 infrastructure components

We develop a cost inventory for construction, retrofitting and installation of 9 specific infrastructure components that support urban adaptation and mitigation efforts. The components listed are all pre-existing pieces of infrastructures used to manage and control climate induced hazards or energy reductions whether at home or the work place, or to provide protection to weather events such as heatwaves or flooding.

Following discussions and agreement among RAMSES consortium partners a wide range of infrastructure categories and components were considered and 9 infrastructure components are presented in this chapter. We selected these as some of the infrastructure components were already covered by RAMSES Work Package (WP) 2 (Acero et al., 2014; Kallaos et al., 2014; Kallaos et al., 2015a) and WP5 (Floater et al., 2014; Costa and Floater, 2015). But also the availability of commercial data, and the suitability for inclusion in climate impact models at the urban scale WP3 and WP 4 (Tapia et al., 2015) were important factors for selecting the infrastructure components. Data was collected through online searches for the 6 infrastructure components that mainly support adaptation efforts and 3 components that mainly support mitigation efforts. We conducted research across 24 countries and in 13 languages, from a varying number of primary data sources, such as direct quotes, bills of quantities, or tender documents, and secondary data sources, like peer reviewed and grey literature.

The infrastructure components that address heat adaptation included in the inventory are air conditioning, mechanical ventilation and solar blinds. The components that protect against flooding (adaptation) that we included in the inventory are permeable paving, green roofs and levees. The components that support mitigation efforts reported here are double glazing, loft insulation and solar panels. Although double glazing and loft insulation support mitigation efforts they also have some rather modest impacts in terms of adaption due to higher cooling demands during heat waves (Gaterell and McEvoy, 2005), but solar panels are exclusively supporting climate change mitigation efforts.

The cost inventories for each component are based on large scale internet searches for genuine real costs of buying, tendering or invoicing the given components. Searches were performed in various periods from February 2015 to February 2016. We used different key words for each infrastructure component in various languages. Data includes costs retrieved from online retailers, websites, home improvement calculators, news articles, and scientific papers. Table 4-1 summarizes search terms, languages and search periods by infrastructure component. The results of these searches were combined with Seneca’s commercial database on small scale infrastructure pricing.
The costs for the levees were researched by contacting consulting companies e.g. Royal Haskoning, JBA, Deltares, Wald-Corbe and many others; investigating published reports and commercial tenders; contacting city representatives and investigating public consultation and planning permissions; contacting national experts (academic colleagues within and outside the RAMSES network) and national and regional agencies e.g. the Environmental Agency and CIRIA in the UK; the Umweltbundesamt and Lander representatives in Germany; and STOWA and Rijkswaterstaat in the Netherlands.

The majority of the cost data was collected for countries where RAMSES case study cities are located (although referring to national, and not local, level prices), as well as for other European and international countries. Some infrastructure components included the cost of installation (capital and labour), others included only the capital costs. All prices are for 2015 (or equivalent) and expressed in euros (€). To convert the data we used the 2014 average exchange rates provided by the Bank of England to express all currencies in Euros (HM Revenue, 2016). For levees we considered the capital costs (installation costs) and not the enabling or maintenance costs. The construction cost are based on real projects that have been built, tenders or online quotes were received and a price inflator using the Bank of England (2016) Inflation Calculator was used to transpose all the costs to the year 2015.

In order to make comparisons for the different pieces of infrastructure (excluding levees) across cities possible we normalised the data using GDP by metropolitan area. We collected data from the Global Metro Monitor (Brookings Institution, 2014), which was available for all but one case study city, and refers to the year 2014. GDP is expressed in US dollars ($) and at constant 2009 prices and purchasing power parity (PPP) rates. Data for Antwerp was not available from the Global Metro Monitor, and was collected from the OECD (2016), referring to the year 2013. It is expressed in US dollars, at constant 2010 prices and PPP rates, and was converted to 2009 prices using the GDP deflator for Belgium (Bluecomics, 2016). GDP values were converted into Euros using the 2009 average exchange rate from the Bank of England.
4.3 Cost inventories for infrastructure components

Our data gathering process and sources to establish an inventory of costs for the specific pieces of infrastructure is summarized in Table 4-2. Consulting varying numbers of primary and secondary data sources we conducted our research across 24 countries and in 13 languages. The number of countries for which there is data for each infrastructure component ranged from one (USA), for solar blinds, to 23 for loft insulation. In terms of adaptation air conditioning was the component with the largest number of countries (14) where data was available.

<table>
<thead>
<tr>
<th>Table 4-2 Summary of data collected for cost inventories</th>
</tr>
</thead>
<tbody>
<tr>
<td>Component</td>
</tr>
<tr>
<td>-----------------------------</td>
</tr>
<tr>
<td>Air conditioning</td>
</tr>
<tr>
<td>Green roofs</td>
</tr>
<tr>
<td>Levees</td>
</tr>
<tr>
<td>Permeable paving</td>
</tr>
<tr>
<td>Solar blinds</td>
</tr>
<tr>
<td>Mechanical ventilation</td>
</tr>
<tr>
<td>Mitigation</td>
</tr>
<tr>
<td>Double glazing</td>
</tr>
<tr>
<td>Loft insulation</td>
</tr>
<tr>
<td>Solar panels</td>
</tr>
</tbody>
</table>

The next subsections investigate and report the cost inventory for each infrastructure components in turn, describing a summary of the results with a focus on the RAMSES case study cities of Antwerp (Belgium), Bilbao (Spain), Bogotá (Colombia), Hyderabad (India), London (United Kingdom), Rio de Janeiro (Brazil), Skopje (Macedonia), and New York City (United States of America). We present the cost inventories for the infrastructure components in alphabetical order but grouped these into adaptation and mitigation efforts.

4.3.1 Air conditioning

Table 4-3 shows the summary of costs of air conditioning in euro per kilowatt of the device power in different countries. This includes different HVAC (Heating, Ventilation and Air Conditioning) systems such as wall-mounted, window mounted and ducted systems. The table presents the mean or average, minimum and maximum cost for ten EU countries and four non-EU countries, where seven of the RAMSES case study cities are located.

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⁹ AT=Austria, BE=Belgium, BR=Brazil, BG=Bulgaria, CO=Colombia, HR=Croatia, DK=Denmark, FI=Finland, FR=France, DE=Germany, IN=India, IE=Ireland, IT=Italy, MKD=Macedonia, NL=Netherlands, PO=Poland, PT=Portugal, RO=Romania, RU=Russia, SG=Singapore, ES=Spain, SE=Sweden, CH=Switzerland, UK=United Kingdom, US=United States
Table 4-3 Installation costs of air conditioning in EU and non-EU countries

<table>
<thead>
<tr>
<th>Country</th>
<th>Average cost (euro/kW)</th>
<th>Minimum cost (euro/kW)</th>
<th>Maximum cost (euro/kW)</th>
<th>RAMSES case study city</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>EU</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Belgium</td>
<td>413</td>
<td>127</td>
<td>820</td>
<td>Antwerp</td>
</tr>
<tr>
<td>France</td>
<td>730</td>
<td>110</td>
<td>1417</td>
<td></td>
</tr>
<tr>
<td>Germany</td>
<td>225</td>
<td>47</td>
<td>1236</td>
<td></td>
</tr>
<tr>
<td>Italy</td>
<td>307</td>
<td>70</td>
<td>664</td>
<td></td>
</tr>
<tr>
<td>Netherlands</td>
<td>446</td>
<td>127</td>
<td>2400</td>
<td></td>
</tr>
<tr>
<td>Poland</td>
<td>180</td>
<td>92</td>
<td>354</td>
<td></td>
</tr>
<tr>
<td>Portugal</td>
<td>292</td>
<td>52</td>
<td>766</td>
<td></td>
</tr>
<tr>
<td>Spain</td>
<td>241</td>
<td>67</td>
<td>569</td>
<td>Bilbao</td>
</tr>
<tr>
<td>Sweden</td>
<td>322</td>
<td>158</td>
<td>939</td>
<td></td>
</tr>
<tr>
<td>UK</td>
<td>244</td>
<td>86</td>
<td>1156</td>
<td>London</td>
</tr>
<tr>
<td><strong>Non-EU</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Brazil</td>
<td>157</td>
<td>88</td>
<td>1502</td>
<td>Rio de Janeiro</td>
</tr>
<tr>
<td>Colombia</td>
<td>147</td>
<td>64</td>
<td>1254</td>
<td>Bogota</td>
</tr>
<tr>
<td>India</td>
<td>104</td>
<td>6</td>
<td>484</td>
<td>Hyderabad</td>
</tr>
<tr>
<td>US</td>
<td>76</td>
<td>49</td>
<td>175</td>
<td>New York City</td>
</tr>
</tbody>
</table>

The costs include both the costs of materials and the labour cost of installation. Variation of costs collected is high for some countries, such as the UK, the Netherlands, Brazil and Colombia. Average costs vary greatly, from a minimum of 76 €/kW in the United States (US) to a maximum of 730 €/kW in France. Figure 4-1 presents a comparison of average costs for each of the 14 available countries.

![Figure 4-1 Average air conditioning cost across countries](image)

Figure 4-2 depicts for each country where a RAMSES case study city is located, the average, minimum, and maximum costs of air conditioning systems. Cities are arranged by order of average costs, from largest to smallest.
Figure 4-2 Costs of air conditioning in RAMSES cities

Figure 4-3 presents again each city’s average cost of installation of air conditioning, in euro per kW, but expressed as a ratio of city GDP per capita for the year 2014. GDP per capita is expressed in 2009 euros and PPP terms, and estimated as set out in Section 4.2. The ratio is multiplied by 100, which results in the following equation.

\[
\frac{\text{Cost (€/kW)}}{\text{GDP}} \times 100
\]

Equation 4-1 Cost to GDP ratio

Cities are arranged by magnitude of total average costs, from largest to smallest, for comparison with Figure 4-2. The city of Rio de Janeiro presents by far the largest cost to GDP ratio (approximately 4.3), followed by Bilbao and Bogota. Relative to their GDP, London and Antwerp have relatively small costs of air conditioning installation, while in terms of total costs they had the second and third largest.

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10 The exception being Antwerp, for which only 2013 GDP was available.
Finally, we look at the relationship between air conditioning installation costs and kW installed. Figure 4-4 presents a scatterplot of total air conditioning installation costs versus kW installed, excluding labour costs. Additionally, three linear and exponential trends for each of the RAMSES core care study cities (Antwerp, Bilbao and London) are presented, along with the corresponding $R^2$ (in coloured boxes for the linear trends). The latter measures how close the data is to the fitted regression line. Costs do not seem to increase linearly with kW installed, particularly for Bilbao and London, as seen by the values of the linear regression’s $R^2$. In particular for the latter, the data fit better an exponential model of costs as a function of kW installed, with costs increasing more for higher power installed.

Figure 4-3 Cost to GDP ratio of air conditioning in RAMSES cities

Figure 4-4 Cost of air conditioning excluding labour
4.3.2 Green roofs

Data on costs of green roofing was collected for six countries and expressed in euros per square meter. Figure 4-5 shows the average cost of green roofing in the countries for which data was available.

![Figure 4-5 Average costs of green roofing by country](image)

Furthermore, some of the collected costs were divided between costs of extensive and intensive green roofs. Extensive green roofs are shallow (3 to 5 inches, or around 7.5 to 12.5 centimetres deep), generally intended to be self-sustaining (requiring low maintenance) that have low construction costs, as they require little to no reinforcement of the existing roofing and fewer components when installed. Intensive green roofs are deeper (7 to 24 or more inches, or around 18 to 36 centimetres deep), require high maintenance and irrigation, and have larger construction costs (Ecolife, 2016; Green Roof Technology, 2016), but absorb more rain and provide more evaporative cooling.

![Figure 4-6 Costs of green roofing by type](image)

Figure 4-6 depicts average costs divided by type of green roof for those countries where they were categorized. The relative difference between intensive and extensive green roofs is highest in Canada and in the USA, and smallest for the UK. This might have implications on the type of green roofs that are most cost-effective in each of the countries.
4.3.3 **Levees**

Levees (embankments or dikes) are an integral part of protecting cities against coastal and river floods and of increased importance in light of climate change (Glendinning et al., 2009a). The averted annual losses from sea level rise for London, Bilbao and Antwerp and estimated construction costs for different levels of protection using the literature (Jonkman et al., 2013) was reported by RAMSES D5.2 (Costa and Floater, 2015), based on analyses by D1.2 (Boettle et al., 2016) which forms the basis of a paper that will be submitted to a peer reviewed journal (Boettle et al., Draft).

Figure 4-7 reports the construction costs of installing fluvial and coastal levees for three European countries from 105 projects from the UK, 18 from Germany and 14 from the Netherlands. Some datasets have only provided average costs (de Grave and Baarse, 2011; Cabinet Office, 2013; LAWA, 2014; Keating et al., 2015) but it can be clearly seen that the construction costs do vary widely. The high average costs in the Netherlands as compared to the other two countries (over 10,000€/m) is due to the high cost for specific restructuring projects that need heightening of existing levees close to urban areas. Once we take these out of the equation the average cost of coastal flood protection in the Netherlands is reduced to some 6,600€/m.

![Average construction costs of levees in selected EU countries](image)

Figure 4-7 Average construction costs of levees in selected EU countries

Local conditions, design, materials and construction methods that are applied will have significant influence on the costs of construction (Vemury et al., 2015). But also local economic factors, such as the costs of labour and materials and the use of equipment as well as differences between countries will have significant impacts (Jonkman et al., 2013). Another important factor is that all 18 projects in Germany, 66 projects described by the Environment Agency and JBA (Keating et al., 2015) and 8 projects we found data for in the UK, are fluvial and river flood protection projects. Thus apart from the protection of London and some coastal projects, which have the highest costs in the UK the vast majority are inland. The costs for the construction of levees (earth filed embankments) in New York also vary substantially depending on location, type of storm surge barrier strategy and construction method is used (Aerts et al., 2013a). But it is clear that these numbers are also significantly higher than for river, or tidal river protections levees.
### Table 4-4 Costs of levees

<table>
<thead>
<tr>
<th>Country</th>
<th>Average cost (€/m)</th>
<th>Min cost (€/m)</th>
<th>Max cost (€/m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Germany</td>
<td>1,057</td>
<td>364</td>
<td>2,445</td>
</tr>
<tr>
<td>Netherlands (Antwerp)</td>
<td>10,041 (966)</td>
<td>924 (924)</td>
<td>26,880 (1,008)</td>
</tr>
<tr>
<td>UK (London)</td>
<td>3,517 (12,085)</td>
<td>328 (8,057)</td>
<td>16,144 (16,114)</td>
</tr>
<tr>
<td>USA (New York)</td>
<td>(17,649)</td>
<td>(9,540)</td>
<td>(25,758)</td>
</tr>
</tbody>
</table>

As can be seen in Table 4-4 the minimum cost of a levee in Germany was 364€/meter, which was the Deich Großkühnau West 2 project. There are large variations even within the same location. As highlighted by the Cabinet Office (2013) costs vary due to distance of site from material sources (on some sites it is possible to source fill material from on-site borrow pits, elsewhere this may not be possible); ease of access to the site; and development in construction techniques to reduce construction costs. Such differences are highlighted by the Dumfries-Whitesands Flood Protection Scheme, which required 1255m long levees and the cost ranged between 328 and 2,375€/meter depending which stretch of levee is constructed. Such differences are explained as one stretch is only 1m high and 195m long, whereas the most expensive stretch needed to be 3.8m high and 71m long. Overall it becomes evident that the provision of coastal levees is significantly more expensive than the costs of constructing fluvial levees.

#### 4.3.4 Mechanical ventilation

Whilst mechanical ventilation is a common component to climate change adaptation, it is normally professionally installed and done on a quote by quote basis, so it was more difficult to find generalized online costs. Nevertheless we collect data for 7 countries, for a maximum of 57 types of mechanical ventilation. The average costs, across all types, by country in euros are presented in Table 4-5. Average costs vary from 695€ for the US and 6300€ for Italy.

### Table 4-5 Average costs of mechanical ventilation

<table>
<thead>
<tr>
<th>Country</th>
<th>Average cost (€)</th>
</tr>
</thead>
<tbody>
<tr>
<td>France</td>
<td>1141.67</td>
</tr>
<tr>
<td>Germany</td>
<td>5619.09</td>
</tr>
<tr>
<td>Italy</td>
<td>6300.00</td>
</tr>
<tr>
<td>Portugal</td>
<td>224.91</td>
</tr>
<tr>
<td>Spain</td>
<td>2000.00</td>
</tr>
<tr>
<td>UK</td>
<td>2984.91</td>
</tr>
<tr>
<td>US</td>
<td>695.22</td>
</tr>
</tbody>
</table>

Because the necessary ventilation system varies with several factors, most of the costs collected were not available in terms of square meter of area ventilated. Nevertheless, we collected a few available in these terms for countries where RAMSES case study cities are located. Figure 4-8 depicts the average cost in the three RAMSES case studies cities where data was collected, in euros per square meter (m²).
The relative ordering of costs does not change when we consider costs (in euros per square meter) divided by GDP per capita. The ratios are of 7.3 for Antwerp, 2.9 for London and 1.9 for New York. This means that, even when considering differences in the countries’ economic power, costs are still highest in Bilbao, and the difference between cities is even more accentuated.

4.3.5 **Permeable paving**

Costs for permeable paving were grouped into asphalt-based, concrete-based or other, and are standardized to euros per square metre (m²). The only countries which had available data for this piece of infrastructure were France, the UK and the US. Table 4-6 presents the average, minimum and maximum cost (in Euros per m²) of the three types of permeable paving as an average over all three countries, and Figure 4-9 Costs of permeable paving by country depicts the average for each country.

<table>
<thead>
<tr>
<th>Type of permeable paving</th>
<th>Average cost (€/m²)</th>
<th>Min cost (€/m²)</th>
<th>Max cost (€/m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Asphalt</td>
<td>21.35</td>
<td>4.06</td>
<td>50.51</td>
</tr>
<tr>
<td>Concrete</td>
<td>51.75</td>
<td>16.23</td>
<td>81.15</td>
</tr>
<tr>
<td>Other</td>
<td>131.57</td>
<td>4.65</td>
<td>284.03</td>
</tr>
</tbody>
</table>

![Figure 4-9 Costs of permeable paving by country](image-url)
4.3.6 Solar blinds

For solar blinds, data was only available in the United States. However, various different sources were found in the US and we categorized the data by level of solar blocking. Table 4-7 presents the average cost in euros per square metre of solar blinds depending on their level of blocking radiation, ranging from 70% to 90% of blocked radiation.

<table>
<thead>
<tr>
<th>Level of solar blocking</th>
<th>Average cost (€/m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>70%</td>
<td>10.40</td>
</tr>
<tr>
<td>80%</td>
<td>12.63</td>
</tr>
<tr>
<td>90%</td>
<td>15.91</td>
</tr>
</tbody>
</table>

4.3.7 Double Glazing

Data for costs of double glazing were collected for ten EU countries and for the US. Table 4-8 presents average costs per square meter (m²), height times width. Average costs per m² are lowest in Romania, followed by the Netherlands and the highest costs are found in Sweden.

<table>
<thead>
<tr>
<th>Country</th>
<th>Average cost (€/m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Belgium/France</td>
<td>94.19</td>
</tr>
<tr>
<td>Denmark</td>
<td>116.77</td>
</tr>
<tr>
<td>France/Switzerland/Belgium</td>
<td>97.57</td>
</tr>
<tr>
<td>Italy</td>
<td>203.06</td>
</tr>
<tr>
<td>Netherlands</td>
<td>62.67</td>
</tr>
<tr>
<td>Poland</td>
<td>76.39</td>
</tr>
<tr>
<td>Romania</td>
<td>14.56</td>
</tr>
<tr>
<td>Spain</td>
<td>223.74</td>
</tr>
<tr>
<td>Sweden</td>
<td>491.81</td>
</tr>
<tr>
<td>UK</td>
<td>237.55</td>
</tr>
<tr>
<td>US</td>
<td>221.93</td>
</tr>
</tbody>
</table>

Figure 4-10 presents a scatterplot of all total costs for different total areas covered, for the four countries in the database where a RAMSES case study city is located. The concentration of values around 1m² is due to the existence of many more data points representing costs per square meter than any other area. Although there is high variability, the total average cost seems to increase at decreasing rates as the total area covered increases.
Loft insulation

Insulation of a loft, attic, or flat roof is an infrastructure component used both for decreasing energy demand and in some cases to decrease inside temperature during heat waves. Studies have shown that, although loft insulation offers modest opportunities for reducing energy demand, when compared to other adaptation efforts, its impact is less dependent on climate change (Gaterell and McEvoy, 2005).

Costs were collected for 24 countries, including several types of loft insulation for each. The average cost in each country ranges from 1.32 €/m$^2$ in Switzerland to 15.38 €/m$^2$ in Russia. Figure 4-11 presents the average costs of loft insulation for each of the countries.
Table 4-9 Average costs of loft insulation by type

<table>
<thead>
<tr>
<th>Main types of loft insulation</th>
<th>Average cost (€/m²)</th>
<th>Min cost (€/m²)</th>
<th>Max cost (€/m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Glass mineral wool</td>
<td>5.13</td>
<td>0.36</td>
<td>43.74</td>
</tr>
<tr>
<td>Polystyrene</td>
<td>8.54</td>
<td>1.31</td>
<td>63.60</td>
</tr>
<tr>
<td>Rockwool</td>
<td>9.81</td>
<td>1.02</td>
<td>377.13</td>
</tr>
</tbody>
</table>

Finally, Figure 4-12 and Figure 4-13 depict the implication of the costs for RAMSES case study cities where data was available. Figure 4-12 indicates that average costs in euros per square meter are highest for the three European cities, but the range of costs collected is also highest here, implying actual costs could be very similar on some occasions.

![Figure 4-12 Costs of loft insulation in RAMSES cities](image)

<table>
<thead>
<tr>
<th>City</th>
<th>Max cost</th>
<th>Min cost</th>
<th>Average cost</th>
</tr>
</thead>
<tbody>
<tr>
<td>Antwerp</td>
<td>38.88</td>
<td>1.59</td>
<td>14.74</td>
</tr>
<tr>
<td>Bilbao</td>
<td>62.10</td>
<td>2.42</td>
<td>10.39</td>
</tr>
<tr>
<td>London</td>
<td>29.31</td>
<td>2.02</td>
<td>7.89</td>
</tr>
<tr>
<td>New York</td>
<td>11.29</td>
<td>2.44</td>
<td>6.33</td>
</tr>
<tr>
<td>Bogota</td>
<td>7.53</td>
<td>3.24</td>
<td>5.13</td>
</tr>
<tr>
<td>Rio De Janeiro</td>
<td>3.58</td>
<td>2.96</td>
<td>3.29</td>
</tr>
</tbody>
</table>

![Figure 4-13 Cost to GDP ratio of loft insulation for selected RAMSES cities](image)

Figure 4-13 presents the cost to GDP ratio (times 100000), with cities organized by order of magnitude of absolute costs, from largest to smallest. Although Antwerp still presents the largest costs when weighted by city GDP, London presents the second to smallest costs, with New York having the smallest.
4.3.9 Solar panels

Data for costs of installation of solar panels was collected for 16 countries and measured in terms of installed power (euro per kW). For three countries (Spain, UK and US) costs were also available measured in terms of installed area (euro per m2). The average cost of installation of solar panels in euro per kW in all 16 countries is depicted in Figure 4-14 Average costs of solar panels across countries. It ranges from a minimum of 0.5€/kW for Germany to a maximum of 4.8€/kW for Estonia.

![Figure 4-14 Average costs of solar panels across countries](image)

The implied cost for RAMSES case study cities is depicted in Figure 4-15. It varies from a minimum of 0.7€/kW for Hyderabad to a maximum of 2.6€/kW for London. Figure 4-16 depicts the cost to GDP ratio for each city, where cities are organized by magnitude of absolute costs, from largest to smallest. The ratio is multiplied by 100,000 for easiness of reading. It can be seen that although London and New York had, respectively, the largest and fourth largest costs in absolute terms, when weighted by their GDP they have the smallest. Hyderabad, on the other hand, has the second largest costs in terms of cost to GDP ratio.

![Figure 4-15 Average costs of solar panels in RAMSES cities](image)
We further investigate how total installation costs vary with installed power (in kW). The results differ between countries. As an example, Figure 4-17 and Figure 4-18 plot the total costs for different levels of power installed in all data points collected respectively for India and the US. While in the former case there seems to be a linear trend, where total costs increase linearly with power installed, for the latter total cost seems, from visual inspection, to be a concave function of the power installed. This means the price increases with kW installed but at decreasing rates, making larger investments cheaper.

![Figure 4-16 Cost to GDP ratio for solar panels in RAMSES cities](image)

![Figure 4-17 Total cost vs power in India](image)
4.4 Implications for RAMSES case study cities and countries

The costs collected were available and presented here to varying degrees for 7 of the 8 RAMSES case study cities.\textsuperscript{11} Data on installation of air conditioning and solar panels was available for all cities. Loft insulation costs were available for 6 of the cities, mechanical ventilation and double glazing costs for 5 of the cities, and costs of green roofs, levees and permeable paving for 3 RAMSES cities. The costs for solar blinds were available only for one of the case study cities. In general our results suggest that data is relatively more available for infrastructure components that support mitigation efforts than adaptation. Table 4-10 and Table 4-11 present respectively for adaptation to heat waves and flooding, a summary of average, minimum, and maximum costs for each piece of infrastructure that support adaptation efforts for which data was collected for each of the 7 RAMSES case study cities.

Table 4-10 Costs of infrastructure components for heat waves in RAMSES cities

<table>
<thead>
<tr>
<th>RAMSES City</th>
<th>Air conditioning</th>
<th>Mechanical ventilation</th>
<th>Solar Blinds</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Min (€/kW)</td>
<td>Max (€/kW)</td>
<td>Min (€/m²)</td>
</tr>
<tr>
<td>Antwerp</td>
<td>127.00</td>
<td>820</td>
<td>NA</td>
</tr>
<tr>
<td>Bilbao</td>
<td>67.00</td>
<td>569</td>
<td>20</td>
</tr>
<tr>
<td>London</td>
<td>86.00</td>
<td>1156</td>
<td>8.27</td>
</tr>
<tr>
<td>Bogota</td>
<td>64.00</td>
<td>1254</td>
<td>NA</td>
</tr>
<tr>
<td>Hyderabad</td>
<td>6.00</td>
<td>484</td>
<td>NA</td>
</tr>
<tr>
<td>New York</td>
<td>49</td>
<td>175</td>
<td>4</td>
</tr>
<tr>
<td>Rio de Janeiro</td>
<td>88</td>
<td>1502</td>
<td>NA</td>
</tr>
</tbody>
</table>

\textsuperscript{11} No data was available for Macedonia (where the city of Skopje is located) for any of the adaptation measures.
Table 4-11 Costs of infrastructure components for flooding in RAMSES cities

<table>
<thead>
<tr>
<th>RAMSES City</th>
<th>Green roofs</th>
<th>Permeable paving</th>
<th>Leveses</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Av (€/m²)</td>
<td>Min (€/m²)</td>
<td>Max (€/m²)</td>
</tr>
<tr>
<td>Antwerp</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Bilbao</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>London</td>
<td>126.59</td>
<td>62.06</td>
<td>248.22</td>
</tr>
<tr>
<td>Bogota</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Hyderabad</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>New York</td>
<td>201.43</td>
<td>92.48</td>
<td>1026.16</td>
</tr>
<tr>
<td>Rio de Janeiro</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
</tbody>
</table>

Table 4-12 presents the summary of average, minimum, and maximum costs of infrastructure components that support climate change mitigation efforts for each case study city where data on these was collected.

<table>
<thead>
<tr>
<th>RAMSES City</th>
<th>Double glazing</th>
<th>Loft insulation</th>
<th>Solar Panels</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Av (€/kW)</td>
<td>Min (€/kW)</td>
<td>Max (€/kW)</td>
</tr>
<tr>
<td>Antwerp</td>
<td>94.1</td>
<td>51.6</td>
<td>178.1</td>
</tr>
<tr>
<td>Bilbao</td>
<td>223.7</td>
<td>101.7</td>
<td>389.9</td>
</tr>
<tr>
<td>London</td>
<td>237.5</td>
<td>15.5</td>
<td>770.7</td>
</tr>
<tr>
<td>Bogota</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Hyderabad</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>New York</td>
<td>221.9</td>
<td>36.9</td>
<td>600.5</td>
</tr>
<tr>
<td>Rio de Janeiro</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
</tbody>
</table>

The tables highlight once again the lack of available data, especially in the context of cities in less developed countries. This situation is particularly striking in what concerns adaptation efforts. While measures and infrastructure components for mitigation have received more attention in the literature, the fact that concerns for adaptation are relatively newer. This means that a higher effort is required in order to estimate the installation costs for providing the necessary infrastructure associated with the protection of the city and its citizen. In terms of the magnitude of costs, our analyses in previous sections indicate that, when measured in terms of cost to GDP ratio, the costs are disproportionally higher in cities with lower GDP per capita.

In order to evaluate each city’s ability to collect and make data available, we investigated the relationship between city GDP per capita and the number of infrastructure components that support adaptation efforts for which data was available. Figure 4-19 plots, for each case study city, the number of components for which data was available, and GDP per capita in constant 2009 euros and PPP rates. A positive relationship seems to be visible, suggesting more data seems to be available for cities with larger economic power. This signals an extra difficulty for cities in developing countries, which face a large bulk of climate change costs, but that might not have the tools necessary available to effectively implement adaptation.
4.5 Conclusions from investigating the infrastructure components

It is expected that adverse impacts, including more frequent and/or severe weather extremes, will increase the need for financial accounting and reporting standards across various infrastructure systems, cities and also companies (Linnenluecke et al., 2015). A wide range of data and detailed inventories for 9 infrastructure components that support climate change adaptation and mitigation efforts were provided in this chapter. But it should be recognised that a narrow focus on purely the quantifiable costs and benefits can bias decisions against the poor, ecosystems, and indeed against future generations whose values may be excluded or understated in such an inventory (IPCC, 2014a). We can confirm the findings by RAMSES D3.1 that the costs of climate change mitigation and adaption do depend on the specific climate impacts targeted (Tapia et al., 2015). And similar to the findings by D5.2 (Costa and Floater, 2015) we can provide only some broad conclusions on the average installation costs for the different components. The quantification of the investment that is needed locally or the benefits or trade-offs to install or construct specific infrastructure components needs to be done in a dedicated assessment that considers e.g. local conditions, designs, raw materials or socio-economic. Nevertheless this chapter does provide some useful benchmark costs for 9 infrastructure components, namely: air conditioning, double glazing, green roofs, levees, loft insulation, mechanical ventilation, permeable paving, solar blinds and solar panels.

Although it is impossible to standardise the costs across the different components, we provide here some general observations. France has the highest installation costs of air conditioning units in the EU, whereas the US has the lowest overall. For mechanical ventilation the US again is the cheapest and Italy has the highest. In contrast, US has the highest installation costs for permeable paving and green roofs. The provision of double glazing in RAMSES countries is the cheapest in the Netherlands and the most expensive in the US and Spain. Loft insulation is most expensive in Belgium and cheapest in Brazil and solar panels are most expensive in the UK and cheapest in India.

The large range and variability of price differences across countries suggests that a range of factors needs to be considered before underlying drivers of costs can be estimated. Examples of such factors could include climate (e.g. countries with warmer climates have higher demand for heat reduction measures which can increase or decrease prices); socio-economic differences in installation (e.g. air conditioning is purchased predominantly by higher earners in lower income countries); or differences in supply costs (e.g. due to availability of domestic natural resources such as sand for levees, or tariff rates on the import of commodities and manufactured products). The variability of prices ranges also suggests that top down assumptions for costs across countries could be misleading.
Where possible we conducted detailed analyses, normalisation and standardisation for the different components across the RAMSES countries in order to position the costs in a local or at national context. In terms of data availability, more cost data was available for infrastructure components that support heat wave adaptation than components that protect against flooding, and for widely used components that address adaptation and mitigation, such as solar panels and air conditioning. Furthermore, our inventory confirms that more data tends to be available for countries with higher economic power. In particular, RAMSES case study cities with higher Gross Domestic Product (GDP) per capita had the largest amount of data available. This has implications in terms of the capacity of each country to compare the cost efficiency of different adaptation possibilities, and highlights the need for transferrable methodologies that minimize the need for detailed economic data, one of the objectives of the RAMSES project. The inventories set out in this chapter will be used in the estimation of aggregate adaptation costs to be included in Deliverable 5.3. The value of the data collected, however, goes beyond the work that will be done within the RAMSES project. This chapter represents an effort to provide a much needed set of standardized data that can be used to evaluate different adaptation policies. This is vital for policy makers to evaluate, compare and prioritize different adaptation options, so as to better allocate resources, a task of increased importance in the context of projected climate change.
5 Assessment and selection methods for Green Urban Infrastructures (GUI)

As described in the DoW (Kropp, 2013) WP1 and T1.3 is conducting selected cases studies and impact approaches for cities and infrastructure systems. We selected Green Urban Infrastructure (GUI) as one of the case studies to investigate how natural infrastructure can support the implementation of climate change strategies (cf. Chapter 2). In RAMSES D2.1 the review of resilient architecture and infrastructure identified and analysed resilience frameworks and corresponding indicators amongst others for GUI (Kallaos et al., 2014). However as is argued by our colleagues the determining indicators or factors are very often not directly measurable, which makes urban adaptation and efficiency strategies difficult to quantify. RAMSES D4.3 (Lobaccaro et al., 2016) investigated the effect GUI can have on adaptation efforts to reduce Urban Heat Island (UHI) and demonstrated that GUI such as tree-lined streets have the potential to reduce the UHI risk locally and in the nearby areas, but not much beyond.

We investigated the contribution GUI can make on mitigation efforts and this chapter, which is based on a paper that was published in January 2016 in the peer-reviewed journal “Environmental Pollution” (Tiwary et al., 2016), we introduce a Performance Indicator (PI). The benefits and trade-offs of GUI in general are described by RAMSES D2.4 (Kallaos et al., 2015a) and specific contributions at the mesoscale and microscale for UHI are described by (Lobaccaro et al., 2016). We present here the methodological framework in developing the PI. As we present a new method in selecting GUI the research has relevance to future RAMSES work such as the stakeholder dialogues (WP9), toolbox development (WP10) and to stimulate urban strategies for transition (WP8). We demonstrate the applicability of the PI through a case study that was conducted in Newcastle upon Tyne (UK). We discuss the relevance of such an approach and its limitations in selecting specific GUI. It is noteworthy that although this chapter reports on streetscape vegetation the PI can be equally used to assess sustainability potential of other GUI such as urban forests, swales, green roofs etc.

5.1 Introduction to streetscape and GUI

Cities, typically through scattered pattern of low-density developments, provide significant potentials for boosting investment in Green Urban Infrastructure (GUI) through managed vegetation along streetscapes i.e. specimens growing along the verge streets, roads or bridleways (LAEC, 2007; Jim and Chen, 2008; Dawe; Ignatieva et al., 2010; Kallaos et al., 2015a). Conventionally, urban greening initiatives had been either ad-hoc or motivated primarily for enhancing amenity values or maintaining biodiversity (Llausàs and Roe, 2012). In cities typically characterized with heavy industrial or traffic activities ‘green belts’ have been integral part of streetscaping (along ring roads and arterial/trunk routes), primarily introduced to mitigate odour, noise and air pollution (Chaulya, 2004; Rao et al., 2004; Pathak et al., 2011). In the last decade however, the concept of GUI has re-defined spatial planning practice (McDonald et al., 2007; Llausàs and Roe, 2012; Bramald et al., 2015), mainly acknowledging its multi-functional inputs to cost-effective, sustainable climate change mitigation including management of carbon (Grimm et al., 2008; Escobedo et al., 2010; Demuzere et al., 2014), air quality (Tiwary et al., 2009; Nowak et al., 2014), and/or energy demands as a source of bioenergy (de Richter et al., 2009); embankment stabilisation (Glendinning et al., 2009b), storm water attenuation (Stovin et al., 2008; Demuzere et al., 2014); energy conservation etc. (Roy et al., 2012; Churkina et al., 2015; Kallaos et al., 2015a).

Street vegetation as part of GUI constitute substantial portion of green space cover in the predominantly built-up inner-city regions globally, with reported tree densities of up to 158 and 300 per km of street respectively in Melbourne, Australia and Guangzhou, China (Kendal et al., 2011). Several cities have developed roadside vegetation management plans, inviting developers and residents to participate in increasing street tree population alongside their long term preservation (LAEC, 2007; Hall et al., 2012; Heidrich et al., 2013). Traditionally, resilience of urban tree population has been largely dependent on species selection to withstand pest infestations, i.e. natural selection (Raupp et al., 2006). Cities witnessed increased non-native diversity of streetscape vegetation as a result of newly introduced
species. This has been further aggravated to increase urban green cover through fast-track programs to plant millions of trees via national and/or international campaigns (Young, 2011; Zhao et al., 2013; Plant for the Planet, 2014). For example, London Plane tree (Platanus acerifolia) are among the most numerous large street and park trees planted in Greater London (UK); about 40% of all trees population in Leicester (UK) is dominated by four broadleaved species (Davies et al., 2011); and roadside coniferous vegetation dominated by nitrogen-tolerant species outside Munich, Germany (Bernhardt-Römermann et al., 2006). The effect of the urban environment on the physiology of the urban plants is an important topic, not only to understand the adaptation of plants to climate change, but also to define the contribution plants can have on climate change mitigation and adaptation (Calfapietra et al., 2015).

There is an urgent need to improve our understanding of the environmental feedbacks of the green urban infrastructure before decisions are made about species selection in developing GUI. The MillionTreesNYC study, for example, aiming to increase New York City’s tree cover by about 20%, developed a planting priority index combining three main indicators: pollution concentration, population density and low canopy cover (Morani et al., 2011). The scope of this study is to develop and evaluate the inherent factors of a high performing GUI, deemed important for sustainable and widespread climate change mitigation and to a smaller extend adaptation. Working with colleagues from Southampton University our research provides a quantitative method to assess the vulnerability, adaptiveness and efficiency of GUI. This chapter consolidates the essential elements to develop a Performance Index (PI) for plants that form GUI (Tiwary et al., 2016). We introduce the PI, which aids decision making by planners/practitioners by providing comparative evaluations on the multitude of green urban infrastructures as part of the streetscape e.g. plant a line of seasonal woody tree biomass vs. perennial shrubs, or develop a high performing vegetation mix, combining sparse line of trees with an understory.

5.2 Developing the Performance Index (PI)

Understanding and improving the environmental performance of street/roadside vegetation comprehensively (trees, shrubs, forbs etc.) has motivated the development and adoption of an index-based framework (Tiwary et al., 2016). Several researchers have expended efforts towards developing performance indicators for specific application of urban trees – for example, towards greenbelt development for pollution alleviation (Prajapati and Tripathi, 2008), for reducing of traffic-generated noise (Pathak et al., 2011), and for more comprehensive evaluation of their ecosystem services and goods from urban forests (Dobbs et al., 2011; Kenney et al., 2011) etc. The PI is conceived in this study as a combination of the following five traits (the first three depicting its multi-functionality and the fourth its resilience) alongside a fifth dimensional trait (essentially as an overriding factor, establishing the fitness for purpose of the species exclusively for street environments):

1. Pollution Flux Potential (PFP) i.e. influence on local-to-regional atmospheric pollutants, comprising of both uptake and release;
2. Carbon Sequestration Potential (CSP) i.e. increased cycling of biogenic carbon;
3. Biomass Energy Potential (BEP) i.e. renewable resource for bioenergy;
4. Environmental Stress Tolerance (EST) i.e. resistance to toxic ambient urban pollutants; and
5. Crown Projection Factor (CPF) i.e. competition for space in the street environment.

Although developing an all-encompassing performance index is deemed incomprehensible (and nearly impossible), the above traits have been considered by us as being important to assess resilient and multi-functionality of GUI. A two-stage approach is adopted – stage one: develop a framework for quantifying and ranking the earmarked traits; stage two: apply this framework to some exemplars, utilizing direct field measurements and/or inventoried literature data to identify the merits and limitations of the Performance Index (PI) approach. A gradation pattern is proposed to rank all the five traits following recent literature (Dobbs et al., 2011; Pathak et al., 2011). The following sub-sections provide overviews of the frameworks used in the case study.
5.2.1 Pollution Flux Potential

The Pollution Flux Potential (PFP) accounts for the aerial interactions of mainly the foliage - for both the dry deposition and release of air pollutants. Urban vegetation have been found effective filters in scavenging gaseous and particulate air pollution (Tiwary et al., 2009), with recent evaluations on the costs associated to avoided health impacts (Nowak et al., 2014). Nearly all plants emit biogenic volatile organic compounds (BVOC) during reproduction, growth, and defence. The BVOCs are emitted by leaves, flowers, and fruits of plants and these compounds can exacerbate photochemical pollution (Calfapietra et al., 2013). A graphical overview of BVOC emissions rates (in micrograms of isoprene or monoterpenes per gram of leaf mass per hour) for a list of popular urban plants species is presented in (Churkina et al., 2015). PFP of a species has been formulated relatively conveniently using the available information on leaf-level processes, as a net effect of annual pollutant deposition (P\text{dep}) and emission (P\text{emit}) weighted by its seasonal leaf cover profile (Equation 5-1). The latter is parameterized as a coupled function of the leaf cover during full foliation (expressed as leaf area index, LAI) and its annual profile (expressed as intra-annual foliage factor, IAL i.e. ratio of no. of months with leaf cover to total no. months in one year). This is aimed to account for the physiological differences attributed to seasonal variations for deciduous and coniferous stands, providing a more realistic PFP.

\[
PFP = \left(1 - \frac{P_{\text{dep}}}{P_{\text{emit}}} \right) \times \text{LAI} \times \text{IAL}
\]

Equation 5-1 Pollution Flux Potential (PFP)

Both \(P_{\text{dep}}\) and \(P_{\text{emit}}\) (expressed as kg yr\(^{-1}\)) is literature derived based on leaf-level activity values of pollutant depositions and emissions or directly acquired from field campaigns. \(P_{\text{dep}}\) includes dry deposition of the following five air pollutants - ozone (O\(_3\)), sulphur dioxide (SO\(_2\)), nitrogen dioxide (NO\(_2\)), carbon monoxide (CO), and particulate matter less than 10\(\mu\)m (PM10). \(P_{\text{emit}}\) includes emissions of isoprene, monoterpenes and other BVOCs (USDA, 2008).

5.2.2 Carbon Sequestration Potential

Vegetation sequester atmospheric carbon in the form of biomass and sequestration potentials vary widely between species depending on their phenology and growth characteristics. Recent evaluations of carbon storage and sequestration by urban trees have been reported (Escobedo et al., 2010; Nowak et al., 2013). The Carbon sequestration potential (CSP, expressed as kg yr\(^{-1}\)) takes into account the capacity of the entire plant to store carbon within woody, long-lasting tissues considering that fine roots and litter have a relatively fast turnover. The carbon sequestered in the soil has been omitted from these estimates owing to inadequate information to date about the carbon fluxes in urban soils for a diverse range of street tree plantations and their disturbances during road works, soil amendments, etc. Various approaches have been adopted to determine the CSP of tree species, one of which is empirical equations based on field scale studies in terms of the total biomass carbon content (Equation 5-2).

\[
\text{CSP} = \text{AGB} \times \text{TBCF} \times \text{C}
\]

Equation 5-2 Carbon sequestration potential (CSP)

Where AGB is Above Ground Biomass (kg yr\(^{-1}\)), TBCF is total biomass conversion factor (1.28), and C is carbon content of dry mass (kg C kg dry mass\(^{-1}\)) (0.5) (Northup et al., 2005).

5.2.3 Biomass Energy Potential

The bio energy potential (BEP, expressed as MJ yr\(^{-1}\)) evaluates the end-of-life use of the biomass – mainly the woody stock from chips, bark and pruning. Recovery of bioenergy, mainly as heat from the combustion of the managed pruning and coppicing of the street vegetation, is obtained from its heating value on a dry basis (BISYPLAN, 2015). Conventionally, this is expressed in terms of either the Higher Heating Value (HHV) or the Lower Heating Value (LHV) (both expressed as MJ kg\(^{-1}\)). The HHV on a
dry basis is related to the typical stoichiometric chemical composition of the biomass (Equation 5-3) following Sagani (2014):

\[ HHV = 0.341 \times C + 1.322 \times H - 0.12 \times (O + N) + 0.0686 \times S - 0.0153 \times Ash \]

**Equation 5-3 Higher Heating Value (HHV)**

Where, \( C, H, O, N, S \) and \( Ash \) denote the corresponding carbon, hydrogen, oxygen, nitrogen, sulphur and ash content, in \% w/w of the bio-fuel. However, since HHV reflects the total amount of heat energy that is available in the fuel, including the energy contained in the water vapour of the exhaust gases, LHV is considered more appropriate representation of the BEP (Bisyplan, 2012), evaluated as a function of HHV (Sagani et al., 2014). This has been weighted by the annual aboveground biomass (AGB) of a stand (kg yr\(^{-1}\), estimated in Section 2.2) to obtain its gross bio energy potential (Equation 5-4).

\[ BEP = LHV \times AGB = \left( HHV - \frac{2.444 \times 8.936 \times H_{dry}}{100} \right) \times AGB \]

**Equation 5-4 Bio Energy Potential (BEP)**

Where, 2.444 (MJ kg\(^{-1}\)) refers to the latent heat of vaporization of water at 25\(^{\circ}\)C, whilst 8.936 (kg) refers to the quantity of water formed by burning 1 kg of hydrogen. \( H_{dry} \) (MJ kg\(^{-1}\)) denotes the hydrogen content of the fuel.

### 5.2.4 Environmental Stress Tolerance

Environmental Stress Tolerance (EST) depicts the resilience of the street vegetation from water stress and pollution damage. Unlike naturally forested or parkland areas street trees are specifically subjected to excessive environmental stresses induced by traffic-generated air and water pollution (Bignal et al., 2008; Churkina et al., 2015), the latter exacerbated from water stress in disturbed/compacted soils typically used in streetscaping (Quigley, 2004). Acute water stress in plants lead to reduction in the leaf chlorophyll content from production of reactive oxygen species (ROS) in the chloroplast (Pathak et al., 2011). On the other hand, such stresses lead to increase in ascorbic acid content as a defensive response in order to protect thylakoid membranes of leaves from oxidative damage under the influence of increased ROS and plants with high leaf pH show greater tolerance against air pollution (Prajapati and Tripathi, 2008). On this basis the EST can be evaluated on the basis of species-specific analyses of four biochemical parameters (Equation 5-5).

\[ EST = \frac{\left( A \times (T + P) \right) + R}{10} \]

**Equation 5-5 Environmental stress tolerance**

Where \( A \) and \( T \) are ascorbic acid the total chlorophyll content of leaf samples respectively (both obtained as mg g\(^{-1}\) of fresh weight), \( P \) is the leaf extract pH and \( R \) is its relative water content (\%).

### 5.2.5 Crown Projection Factor

The Crown Projection Factor (CPF, expressed as m\(^2\)) is considered as an overriding metrics in evaluation of the fitness for purpose of the species exclusively for street environments. This is to overcome potential competition between the road space and the kerbside vegetation, with negative feedbacks to both the air and the soil environments. Recent studies have reported large street trees as obstacles to airflow i.e. hampering the mixing of pollutants in poorly ventilated areas close to streets and reduced air exchange with the above-roof ambient environment (Gromke and Ruck, 2009; Wania et al., 2012) and deep-rooted species damaging the road fabric (Randrup et al., 2001). The canopy spread of a species, commonly expressed in terms of the canopy projection area in the arboriculture literature, has been found to be
directly proportional to its diameter at breast height (typically for DBH < 100 cm; Shimano, 1997), the CPF has been estimated as a coupled function of DBH and the stand height, H (Equation 5-6) and thus it is not needed to report these results in detail in the following chapter.

\[ CPF = DBH \times H \]

Equation 5-6 Crown protection factor

5.3 Testing the Performance Index (PI)

5.3.1 Site description and species selection

The case study was located on an area spanning approximately 250m×200m adjacent to a busy road network (Figure 5-1), connecting the suburbs to Newcastle upon Tyne city Center, UK (54.979°N, 1.6111°W). The site is close to Newcastle University and its laboratories, which made the field work and transport of the equipment easy to manage.

![Figure 5-1 Site extent (image taken from Google Maps)](image)

An initial visual assessment of species abundance, proximity to the road and suitability for assessment was carried out to draw a shortlist of fifteen species, which comprised a good mix of deciduous and evergreen trees and shrubs in the UK (Table 5-1). Shrubs were specifically included towards developing a holistic assessment framework for the entire roadside vegetation fabric. Inclusion of shrubs and forbs have been recommended in the literature for a better understanding of the full suite of multi-functionality of the urban ecosystems (Dobbs et al., 2011).
Table 5-1 GUI and street vegetation species used in evaluation

<table>
<thead>
<tr>
<th>Species</th>
<th>Stand height (m)</th>
<th>DBH (cm)</th>
<th>Type</th>
<th>LAI</th>
<th>IAL</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Trees</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Horsechestnut (Aesculus hippocastanum)</td>
<td>16.70</td>
<td>63.90</td>
<td>D</td>
<td>5.55</td>
<td>0.58</td>
</tr>
<tr>
<td>Sycamore Maple (Acer pseudoplatanus)</td>
<td>9.37</td>
<td>32.44</td>
<td>D</td>
<td>2.76</td>
<td>0.75</td>
</tr>
<tr>
<td>Hornbeam (Carpinus betulus)</td>
<td>12.57</td>
<td>7.15</td>
<td>D</td>
<td>2.03</td>
<td>0.75</td>
</tr>
<tr>
<td>Turkish Hazel (Corylus colurna)</td>
<td>13.03</td>
<td>14.73</td>
<td>D</td>
<td>3.02</td>
<td>0.60</td>
</tr>
<tr>
<td>Beech (Fagus sylvatica)</td>
<td>19.5</td>
<td>99.10</td>
<td>D</td>
<td>6.12</td>
<td>0.75</td>
</tr>
<tr>
<td>Ash (Fraxinus pensylvanica)</td>
<td>11.84</td>
<td>24.39</td>
<td>D</td>
<td>4.11</td>
<td>0.58</td>
</tr>
<tr>
<td>Sweet gum (Liquidambar styraciflua)</td>
<td>15.85</td>
<td>30.50</td>
<td>D</td>
<td>3.62</td>
<td>0.67</td>
</tr>
<tr>
<td>London Plane (Platanus × acerifolia)</td>
<td>20.51</td>
<td>63.85</td>
<td>D</td>
<td>2.40</td>
<td>0.67</td>
</tr>
<tr>
<td>SRC Willow (Salix spp.)</td>
<td>10.17</td>
<td>14.13</td>
<td>D</td>
<td>2.31</td>
<td>0.75</td>
</tr>
<tr>
<td>Lime – Littleleaf Linden (Tilia cordata)</td>
<td>7.64</td>
<td>24.32</td>
<td>D</td>
<td>3.87</td>
<td>0.60</td>
</tr>
<tr>
<td>Norway Spruce (Picea abies)</td>
<td>13.4</td>
<td>44.4</td>
<td>E</td>
<td>9.80</td>
<td>1.00</td>
</tr>
<tr>
<td><strong>Shrubs</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Black Cherry (Prunus serotina)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Berberis (Berberis stenophylla)</td>
<td>3.27</td>
<td>12.23</td>
<td>D</td>
<td>2.44</td>
<td>0.60</td>
</tr>
<tr>
<td>Larustinus (Viburnum tinus)</td>
<td>2.25</td>
<td>7.25</td>
<td>E</td>
<td>3.27</td>
<td>1.00</td>
</tr>
<tr>
<td>Mahonia (Mahonia japonica)</td>
<td>5.20</td>
<td>8.3</td>
<td>E</td>
<td>3.52</td>
<td>1.00</td>
</tr>
<tr>
<td>Horsechestnut (Aesculus hippocastanum)</td>
<td>1.90</td>
<td>5.74</td>
<td>E</td>
<td>2.92</td>
<td>1.00</td>
</tr>
</tbody>
</table>

DBH- Diameter at breast height, LAI- Leaf area index, D- deciduous; E- evergreen; IAL- Intra-annual leaf cover (Twary et al., 2016)

5.3.2 Sampling and analysis

All sampling was performed within 100 m of the verge of the main road since literature evidence suggests strongest effects of traffic-generated pollutants in the first 50-100 m from a road (Bignal et al., 2008); with particulates decreasing in concentration more rapidly than gaseous constituents, and gases with a high deposition velocity (such as HNO2 and NH3) decreasing more rapidly than those with a lower deposition velocity, such as NO and NO2 (Truscott et al., 2005). The earmarked traits for the exemplar species were evaluated using a combination of experiments and literature survey (Table 5-3).

5.3.3 Pollution Flux Potential

Inventory data in the i-Tree (UFORE) model (Nowak et al., 2006; USDA, 2008) have been used for both $P_{dep}$ and $P_{emi}$. This approach overcame the complexities in simultaneous, long-term measurement of pollutant fluxes in busy urban street environments. For $P_{dep}$ validation, nitrogen concentrations were used as proxy given the site was close to heavy traffic activity. The nitrogen analysis was performed following a method adapted from Bignal et al., 2008. For $P_{emi}$ validation, isoprene concentrations have been used as proxy, estimated for UK-specific inventoried leaf-level emissions data (Guenther, 2013).

5.3.4 Carbon Sequestration Potential

Within the study area all trees have been inventoried and structural data measured, i.e. diameter at breast height, height, crown depth, crown wideness, health status of the plant, and crown exposure to light. For each species, its CSP has been considered directly proportional to its AGB (Equation 5-2), the latter expressed as a function of its stand height and the DBH using empirical biomass equation (Table 5-1). The majority of empirical biomass equations are acquired from the literature on European conditions. For example, both Berberries and Larustinus have been generalized using empirical biomass equation for Mahonia. As estimated biomass on the basis of empirical equations is generally found to be higher than field observed values, all outputs were multiplied by a compensatory adjustment factor of 0.8 following (Nowak et al., 1994). Similar to the i-Tree approach, the total biomass estimates were further multiplied by biomass adjustment factor (ranges from 0-1) to adjust for the tree condition as follows: fair to excellent condition – 1, poor condition – 0.76, critical condition – 0.42, dying – 0.15, dead – 0.

5.3.5 Biomass Energy Potential

For estimating the BEP the required constituent chemical composition of woody biofuels - C, H, O, N, S and Ash (see 5.2.3) of the selected species typically representative of temperate climes in Europe and North America were acquired from literature survey (Obernberger et al., 2006; Jaya Shankar et al., 2006).
2011) as described in Table 5-2. Those species which have not been exclusively listed in the literature were approximated as typical values of the following categories – virgin wood thinning (coniferous or deciduous wood/logging residues), wood chips, short rotation coppice pruning.

### 5.3.6 Environmental Stress Tolerance

In order to estimate the EST a sampling protocol was adapted to ensure that the species were subjected to similar stress environments i.e. exposure to traffic air pollutants, soil conditions and insolation levels, and negligible spatial heterogeneity. This was considered since environmental factors like soil, rainfall, temperature are important parameters influencing the pollution tolerance of vegetation (Mickler et al., 2003). Ascorbic acid content of leaf samples was estimated (Queval and Noctor, 2007); total chlorophyll content of the leaves was estimated (Liu and Ding, 2008); leaf pH was determined (Prajapati and Tripathi, 2008) and relative water content was calculated (Pathak et al., 2011), which served as a measure of plant stress from exposure to pollutants. Standard protocols and formulations for sampling and analysis of the four constituent parameters are described in Table 5-2.

### Table 5-2 Parameters and methods to estimate multi-functionality and resilience traits

<table>
<thead>
<tr>
<th>Trait Multi-functionality</th>
<th>Constituent parameter</th>
<th>Method</th>
<th>Literature</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pollutant flux potential (PFP)</td>
<td>Leaf area index (LAI) a</td>
<td>Invented literature data</td>
<td>(USDA, 2008)</td>
</tr>
<tr>
<td>Pollutant deposition (PEp) (g yr⁻¹)</td>
<td>Field survey</td>
<td>(Nowak et al., 2006); USDA (2008)</td>
<td></td>
</tr>
<tr>
<td>Pollutant emission (PEe) (g yr⁻¹)</td>
<td>Estimated as annual average total emission of isoprene, monoterpenes, and other VOCs</td>
<td>USDA (2008)</td>
<td></td>
</tr>
<tr>
<td>Carbon sequestration potential (CSP) (kg yr⁻¹)</td>
<td>Diameter at breast height (DBH) b</td>
<td>Field survey</td>
<td>Various (Tiwary et al., 2016)</td>
</tr>
<tr>
<td>Carbon sequestration potential (CSP) (kg yr⁻¹)</td>
<td>Height of crown base (m)</td>
<td>Field survey</td>
<td>(Tiwary et al., 2016)</td>
</tr>
<tr>
<td>Biomass energy potential (BEP) (MJ yr⁻¹)</td>
<td>Above Ground Biomass (AGB) a (kg yr⁻¹)</td>
<td>Estimated using DBH and stand height data in empirical biomass equations.</td>
<td>(Obernberger et al., 2006)</td>
</tr>
<tr>
<td>Biomass energy potential (BEP) (MJ yr⁻¹)</td>
<td>Heating values a (MJ kg⁻¹)</td>
<td>Obtained from heating value of tree biomass on a dry basis, mainly the woody stock from chips, bark and pruning using literature data (see Section 5.2).</td>
<td>(Sagani, 2014; BISYPLAN, 2015)</td>
</tr>
</tbody>
</table>
| Resilience | Environmental stress tolerance (EST) (mg g⁻¹ fresh weight) | Leaf Ascorbic acid content a | Determined from spectrophotometric analysis of supernatant samples obtained from snap-frozen leaf discs using the formula: 

\[ \frac{E_{\text{e}} - E_{\text{b}} - E_{\text{c}}} {W \times 100} \]

where V= volume extract, W= weight of the leaf sample (g), and E₀, E₁, and E₂ are optical densities of blank sample, plant sample and sample with ascorbic acid respectively. | (Prajapati and Tripathi, 2008) |
| Resilience | Total chlorophyll content a (mg g⁻¹) | Determined from spectrophotometric analysis of optical densities of solutions of leaf pigment extracts (obtained in dark to avoid photo-oxidation of pigments) at 645 and 663nm wavelengths (D₆₄₅ and D₆₆₃ respectively) using the formula: 1.62 (D₆₄₅)+0.64 (D₆₆₃) | (Prajapati and Tripathi, 2008) |
| Resilience | Leaf pH a | Determined using a digital pH meter from supernatant samples of crushed and homogenized 0.5 g of leaf. | (Prajapati and Tripathi, 2008) |
| Resilience | Relative water content b (%) | Calculated from leaf weight (LW) using formula: 

\[ \text{RWC} = \frac{\text{LW}_{\text{fresh}} - \text{LW}_{\text{dry}}} {\text{LW}_{\text{fresh}}} \times 100 \]

| Pathak et al., 2011 |

* *representative estimates based on literature data b direct field measurements (based on Tiwary et al. 2016)
Estimating EST and conducting a long-term sampling campaign for all the species studied over different seasons was considered ambitious and futile, mainly owing to the difficulty in associating environmental stressors with the evergreens during no-leaf periods of deciduous species. As a substitute, we considered it appropriate to set the start of the spring foliation season for the deciduous species as the benchmark for representative estimation of the EST. Thereafter, field sampling of all the constituent parameters for the studied species were obtained in three stages (late-spring, mid-summer, early-autumn), followed by lab analyses.

5.3.7 Implications and potential uses of the Performance Index (PI)

It is interesting to note that different combinations of the constituent traits have contributed to similar Performance Index (PI) outputs (Table 5-3), thereby showing the merits of this comprehensive evaluation framework. For example, Horsechestnut, Beech, Sweetgum and London Plane have typical scores not considered suitable for streetscape vegetation, however, there are marked differences between their contributory constituent traits. Beech, Horsechestnut and London Plane have strong contributions to all four out of the five traits, except for CPF, which undermined their suitability for streetscape vegetation. Whereas, the PI of Sweetgum is completely different - despite having favourable CPF it is not considered favourable owing to its poor PFP (being high BVOC emitter) and lower CSP, BEP and EST scores. Likewise, it is also interesting to note the marked variations in the contributory traits for some of the high performing species. Apart from the majority of them have consistently high CPF scores, which was key determinant for their streetscape worthiness, SRC-Willow has strong CSP and BEP scores, whereas Ash and Norway Spruce have relatively higher PFP and EST scores and very modest CSP and BEP scores. In case of shrubs the PI scores were dominated by their high CPF and modest PFP and EST. The latter two are typical for the evergreen shrubs and considered vital traits for ensuring their suitability as streetscape vegetation as part of GUI.

Overall, among trees, Norway spruce (evergreen species) appears to be the most favourable for streetscaping, with a balanced share of all the evaluated traits. This is followed by Maple, Hornbeam, Ash, SRC Willow, and Lime. Among shrubs there is obvious winner, and all of them are considered favourable for streetscaping. On the other hand, deciduous species such as Horsechestnut, Sweet gum and London Plane have poor overall performance values. This is important to note since currently these species are abundantly utilized as street vegetation in temperate continental climate. The case of Beech is unique – it does score high on its multi-functionality traits so definitely is a high-performing species overall for general urban planting (e.g. parklands, greenspace, woodlands, etc.), but it does not seem favourable for the street environments, solely owing to its unfavourable CPF score. Based on inference drawn from descriptive statistics the evergreen species of trees and shrubs are found to be more favourable for streetscaping.
The proposed PI method brings together the essential components of building GUI and a high-performing streetscaping vegetation. However, the PI is an indicative figure and should not be interpreted as absolute values. In no way should it be treated as a ‘one-size-fits-all’ blueprint for urban vegetation in general. Although also important to some Local Authorities in the UK (Heidrich et al., 2013) this approach is shy of being considered a comprehensive metrics, in particular lacking supporting information on issues of storm water run-off/ flood-risk mitigation and resilience therefrom. Also, we acknowledge the use of inventoried data while evaluating the constituent traits of the PI could be over- or under-estimating the indicator values. Albeit, the inventory generated from the i-Tree UFORE model is the most extensive publicly-available dataset thus far (USDA, 2008), and as presented in the demonstration case study (Section 5.3), it facilitates screening level assessments to explore the trends without excessive dependence on the experimental resources. Nevertheless, richer evaluation of GUI and streetscape should follow representative evaluation of the constituent traits using our methodology (PI). This could also involve detailed analyses of site-specific samples corresponding to the study area’s tree species, climate, seasonality and socio-political context. It is also noteworthy that the units of the traits are to be strictly adhered to for consistency in allocation of representative grading score, failing which will yield anomalous PI score.

We acknowledge the rankings applied to convert the finite estimates for different traits are subjective but care has been taken to keep the weightings uniform, reflecting the information and the recommendations on maximum and minimum values for these traits in the recent literature. It is noteworthy that this study only evaluated the efficiencies from direct energy recovery via biomass (in terms of calorific value). A more holistic evaluation warrants inclusion of nutrient recovery via

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12 The basis of the constituent traits indicating multi-functionality and resilience on an annual basis. Descriptive statistics at the end provide an overall comparison between deciduous and evergreen species for more detail see Appendices of Tiwary et al. (2016).
composting and/or advanced bio refinery processes, which was considered beyond the scope of our assessment. Lateral assessment of roadside vegetation as scavengers of nutrients, could also be twinned towards promoting an innovative street vegetation regime, dominated by species with low BVOC emissions, but at the same time with accelerated response to N-deposition in terms of enhanced growth. Such managed street environments would enhance nutrient utilization capacity in a closed-system, further boosting their PI through positive contributions.

Another important limitation of the proposed PI approach, especially relevant for temperate continental landscapes, is its reliance on an abstract street vegetation environment with a steady-state foliage profile, rather than incorporating a dynamic street environment with varying foliage profile (spring-summer: predominantly sun-lit with optimal foliage performance; autumn-winter: predominantly over-casted or snow-laden with underperforming foliage) and its corresponding phytological responses to the different seasons. This issue affects both the deciduous and the evergreen species, albeit it has more contrasting responses from the cyclic foliation and defoliation of the deciduous species. We envisage that such limitation may not be fully overcome, but should considered in future studies by adequately accounting for the foliage and the seasonal dynamics.

Our study demonstrates that the Performance Index (PI) can be applied in real conditions. It provides a useful assessment and selection method that informs future RAMSES work such as the transition alternatives testing (Task 8.3) and toolbox on climate change adaptation and sustainability and training (Task 10.1 and 10.2) as it helps city planners to select the most suitable species in order to maximize the mitigation and adaptation potential of GUI vegetation. The PI consolidates the essential elements of resilience to develop a PI for street plantation, mainly aiming to maximize their service to the urban community while ensuring their prolonged existence. The premise of the PI approach is that the tree must be well-suited to the specific growing conditions and resilient to threats from pests, drought, storms, etc., otherwise functional performance is moot. The PI presented in this chapter can have important implications for developing more resilient street plantation infrastructure, specifically in the context of scattered urbanization pattern with low-density development, commonly witnessed in urban and peri-urban regions.
6 Cities strategies and Electric Vehicles (EV)

Cities across Europe are planning to mitigate and adapt to climate change and increase their efficiencies (see Chapters 2 and 8 of this report). Physical typology and its impacts on climate change mitigation and adaptation have been described by RAMSES D2.2 (Acero et al., 2014), which focused mainly on the impacts climate change can have on the transportation system. D2.3 showed for the RAMSES cities the challenges and opportunities of the transport infrastructure (Kallaos et al., 2015b) and as discussed by RAMSES D2.1 (Kallaos et al., 2014) the current decarbonising transport strategies rely heavily on electric mobility. One city component that is deemed to help reducing CO2 emissions in cities significantly are the widespread uptake of Electric Vehicles (EV). But are cities responding to such potential uptake and consider the shift of the infrastructure and how are these shifts incorporated in existing climate change mitigation strategies?

This chapter, which is a shorter version of a paper that we aim to submit in May 2016 to a peer-reviewed Journal (Heidrich, Draft), investigates the effectiveness of mitigation strategies from 30 UK Cities (cf. Chapter 2) to encourage the uptake of Electric Vehicles (EV). As it was highlighted in the DoW such uptake requires the provision of public charging points and supporting infrastructure in order to help reduce carbon emissions (Kropp, 2013). We found that 13 out of 30 Urban Audit (UA) cities (cf. Chapter 2) mention EV as part of their climate change mitigation strategy. Yet, there is no statistical difference in the infrastructure provision i.e. number of charging points or number of registered EV for the cities that have EV as part of their climate change mitigation strategy and those that do not. We also evaluated carbon outputs and its change if current vehicles would be steadily replaced with new vehicles, of which a certain proportion. We show that EV, when compared to an efficient ICE (Internal Combustion Engine), might not be the magic bullet to achieve the needed reductions by 2020 or 2027. The research has relevance on future policies of transition (RAMSES Task 7.3) as it shows that combined effort of increasing efficiencies and provide a mix of transport modes from EV to biofuels is needed and cities need to improve their climate change mitigation strategies and make sure they implement the necessary infrastructure systems.

6.1 Introduction to emissions targets, EV and infrastructure

Urban areas are responsible for some 70% of global energy related carbon emissions (IEA, 2008). Today 54% world’s population live in urban areas which is anticipated to increase to 66% by the year 2050 (United Nations, 2015) and there is a recognised and urgent need to concentrate on cities and their sustainable transport strategies for dealing with the challenges (and opportunities) climate change may bring (Banister, 2011; Hickman and Banister, 2014). The IPCC commitment by the UK government is to achieve a 60% carbon reduction from the transport sector by 2050 (IPCC, 2007b; IPCC, 2007a). A self-reported survey of 36 megacities demonstrated that cities believe that they have the power and opportunities to take action implementing over 919 actions in the transport sector such as providing highly efficient buses (e.g. improved fuel economy or using clean fuel), dedicated cycling lanes and many other actions that help to mitigate climate change (ARUP, 2011). Many cities take leadership roles in fostering a modal shift e.g. demand management and other tools available to tackle their transport problems. It is widely recognised that there is a need for an effective coordination of capacity and capability to plan and implement systemic changes in cities (Hodson and Marvin, 2010). It has been argued that in the US such changes are still not accelerated as the climate change problem is not yet perceived large enough, understood or well-aligned to stimulate a full reorientation (Penna and Geels, 2015). Nevertheless UK cities increasingly attempt to reduce the carbon emissions from this sector by reducing the reliance on personal internal combustion engine (ICE) vehicles, but there is considerable variation between cities (Banister, 2011; Hickman and Banister, 2014).

City authorities have tried different transport strategies such as low emission zones, parking restrictions, road user charging (Blythe, 2005), travel plans, trip substitution and encouraging modal shift away from carbon intensive transport modes (Santos et al., 2010; Moriarty and Honnery, 2013; Eurostat, 2015). As reported by Banister (2011) many local governments are taking leadership role in addressing the transport problems as they relate to carbon emissions, but the ones reported by Banister are world
megacities and even amongst those considerable variations of strategies and implementation status exist. Whilst there is a substantial body of work about different climate mitigation strategies for transport, not many have considered the impact urban climate change strategies have on the uptake of Electric Vehicles.

This chapter reports on the impact local climate change mitigation strategies in the UK have on the electric vehicle (EV) uptake. We analyse the relationships between reduction targets and mitigation strategies. The chapter shows the impact required from future climate change strategy which can serve as a useful guideline for future developments of city components such as the charging infrastructure needed for EV. To achieve this aim this chapter addresses the following objectives:

1. Report on climate change mitigation plans published by 30 UK cities and their impact on EV,

2. Predict current and future uptake of EV and infrastructure to meet the UK’s climate change targets

3. Recommend actions for cities to promote EV and infrastructure development

A range of studies have investigated how national incentives and policies can increase EV uptake (Gardner et al., 2013; Tran et al., 2013; Bohnsack et al., 2014; Pasaoglu et al., 2014), but little is known if and how local strategies do impact on EV usage and its supporting infrastructure. Or how does such changes influence emissions and carbon reduction targets? In 2012 the transport sector accounted for 502 MtCO$_2$e (Million Tonnes of Carbon Dioxide equivalents) and contributed 26% to the final energy consumption across EU-28 countries (Eurostat, 2015). Notably, it is the only sector which has seen a rise of GHG emissions compared to 1990 levels and the EU has developed a range of policies in order to mitigate the effects from transport on climate change such as CO$_2$ emission limits for new cars and vans; CO$_2$ labelling for cars; improve fuel quality and promote biofuels. The actions or inactions by central, regional and local governments, as well as companies and society as a whole, can have profound effects on the climate. Cities across Europe are introducing mitigation strategies to tackle the cause of climate change (Reckien et al., 2015; Heidrich et al., 2016). The European Parliament commits its member states to an ambitious climate change strategy reducing GHG emissions and energy consumption by at least 20% by 2020 using a 1990 baseline (European Parliament, 2009). Although there is evidence from across Europe that most national strategies lack the implementation and evaluation process (Biesbroek et al., 2010), but as discussed in Chapter 2 the UK set the pace providing legislative and regulatory frameworks to tackle climate change mitigation (Reckien et al., 2015; Heidrich et al., 2016).

The Climate Change Act (AoP, 2008) placed a duty onto the country to ensure that UK net carbon account for the year 2050 is at least 80% lower than the 1990 baseline. The Act aims to improve carbon management and help the transition towards a low carbon economy in the UK; and to demonstrate strong UK leadership internationally, signalling that the Government and the Country as a whole are committed to reducing emissions. While the domestic greenhouse gas emissions were 28% lower in 2009 compared to 1990 levels, transport emissions remained nearly constant i.e. 122.2 MtCO$_2$e in 2009 compared to 122.1 MtCO$_2$e in 1990 (HM Government, 2011). In order to make road transport more sustainable, the UK Government has promoted the uptake of ultra-low carbon vehicles such as electric cars, hydrogen powered vehicles and biofuels (HM Government, 2011). The UK Government has announced a £37 million investment into public recharging infrastructure at train stations, on public sector estate and on-street and rapid charging networks (Office for Low Emission Vehicles, 2013b; Office for Low Emission Vehicles, 2013a). Central Governmental policies strongly influence the selection of climate change mitigation measures at the city level (Bulkeley and Kern, 2006; Biesbroek et al., 2009; Heidrich et al., 2016). There is an urgent need to provide some quantitative and qualitative evidence to show if cities actively design and implement climate change strategies, or if they just pay lip service by linking climate change strategies to other issues. Furthermore, transport planning is often seen in isolation and often takes into account fuel costs, population growth and rising demand, but not whether the general public wants to adopt the new technologies (Heidrich and Tiwary, 2013; Harvey et al., 2014; Guy et al., 2015). Yet, the transition to a low-carbon transport system is slow.
6.2 Investigating mitigation strategies and EV registration

To facilitate the analysis of mitigation efforts, the climate change strategies, plans and programmes were collected at the city level, i.e. the city is defined by its administrative and/or political boundaries and can be referred to as an Urban Area. Cities (urban areas) were selected following the Urban Audit methodology (Eurostat, 2010; Schwarz, 2010; Morais and Camanho, 2011). The Urban Audit (cf. Chapter 2) aims to provide a balanced and representative sample of European cities (Eurostat, 2007). The Urban Audit lists 30 UK cities and urban areas that are deemed a good representation of the UK as a whole and we included all these cities in our research (Figure 6-1).

![Figure 6-1 Selection of UK cities analysed](image)

We collected and analysed published climate change strategies, plans and programmes (documents) retrieved from the website and/or by contacting the city directly (Heidrich et al., 2013). Although all efforts were made to retrieve relevant documents but it is inevitable that some may be missing. Nevertheless all cities and their representatives provided information and the data gathering process was completed in April 2015, which ensured that the impact of the strategies can be assessed. All 30 cities acknowledge climate change being a threat and that their city is tackling this issue by adapting and mitigating with various levels of planning and success. In the UK cities are part of larger Metropolitan, District and County Councils and some cities do refer to regional strategies. For example Stoke on Trent Council does refer to the “South Staffordshire Council Climate Change Strategy” (South Staffordshire Council, 2008) and Gravesham Council to the “Kent’s Adaptation Plan Action Plan 2011-13” (Kent County Council, 2011).

Car ownership data and household composition data were collected from the Office for National Statistics for England and Scotland the National Records of Scotland for Scottish cities. National travel data were used from the DfT’s National Travel Survey. In order to evaluate the effectiveness of climate
change strategies, the number of charging points, the proportion of EV registered and the relative change in registered EV were analysed for the cities which have an EV strategy and those cities who do not. For the relative change in registered EV, a sub-set of the data was used due to the difficulty in obtaining up to date statistics on EV registration. To understand the impact of differing penetration rates on the total carbon output for private transport, we evaluated current outputs and assessed how these could change under different scenarios.

6.3 Results from analysing city strategies and EV uptake to meet targets

6.3.1 Cities climate change strategies

Climate change strategies, plans and programmes from 30 UK Urban Audit cities were gathered and analysed. The cities represent a population of around 17,300,000, including two Welsh cities (Wrexham and Cardiff), three Scottish (Aberdeen, Edinburgh and Glasgow), two cities from Northern Ireland (Belfast and Derry) and 23 English cities. By far the largest city (if indeed it could be referred to as one city) is London with a population of 7.6 million and the city with the smallest is Stevenage with a population 81,000 (Office for National Statistics, 2011). The greater area of London is most densely populated (4,687.6 residents per km$^2$) and Wrexham the least densely populated city with 257 residents per km$^2$ in 2006 (Eurostat, 2010; Eurostat, 2011). The 8 largest economies (outside London) in England are referred to as Core Cities (Birmingham, Bristol, Leeds, Liverpool, Manchester, Newcastle, Nottingham and Sheffield). These cities, forming the economic and urban cores of their surrounding areas, are major centres of regional and national economic growth (Champion and Townsend, 2011), are all assessed.

Of the 30 UK Urban Audit cities, 28 have published climate change strategies, plans and programmes (documents) outlining how they will tackle climate change mitigation. Derry-Londonderry (Northern Ireland) and Wrexham (Wales) are at the start of this process and have not yet published an official policy document (Heidrich et al., 2013). Overlaps between different strategies, plans and programmes within one city are inevitable. In total 307 documents were provided by the cities. Based on an assessment of suitability for analysis i.e. strategies that state climate change in its title or abstract were selected and 52 documents were analysed in detail. The documents are published at various dates and by different departments, for example, the Climate Change (CC) action programme for Aberdeen is the oldest ‘live’ programme, published in 2002 (Aberdeen City Council, 2002). The mitigation and adaptation strategies for London underwent various stages of consultation over recent years and were finally approved and published in October 2011. Out of the 52 documents, 18 defined the scope as the activities that are controlled by the council and 32 are covering activities across the council i.e. household, industry and business activities. Only documents from Gravesham and Stoke have not stated the scope of the strategy i.e. if the strategy is for the councils own operation only or if it does cover households, industry etc.

Out of the 52 strategies, plans and programmes provided by the 28 cities, 49 address climate change mitigation specifically. Climate change strategies and implementation programmes that address energy (e.g. savings, efficiency and renewable sources) are often economically beneficial, improve energy security and reduce emissions (Kousky and Schneider, 2003; Bulkeley and Kern, 2006; Hunt and Watkiss, 2011; Wende et al., 2012; Heidrich and Tiwary, 2013). Therefore, not surprisingly, all 49 mitigation documents and all 28 cities aim to make energy savings and improve energy efficiencies. A closer investigation shows that almost half (45%) of the documents analysed (and 68% of the cities) state that they want to provide an assessment report to identify mitigation measures that might be needed and feasible within their city. It becomes clear that such an assessment would probably highlight duplications or omissions as is the case in the previous example. Some cities may already have completed an assessment report (e.g. Cambridge, London) and therefore their mitigation strategy is already grounded on such an assessment and thus not referring to such a measure.
6.3.2 Cities strategies, transport and incentives for EV

Transport is listed in 45 (92%) documents by 26 (93%) of the cities are aiming to mitigate climate change by improving transportation (Figure 6-2). Transport measures proposed are wide ranging from providing green travel plan for its staff (Edinburgh City Council, 2007), introducing flexible working hours and low carbon vehicle fleet (Birmingham City Council, 2010) to developing a specific project such as the Bristol Rapid Transit Project (Bristol City Council, 2010) and supporting electric vehicles (either supporting infrastructure or actual vehicles) as mentioned by 46% of cities and 33% of the documents. With regards to electric cars 12 of the 25 Local Authorities had strategies promoting electric vehicles in one shape or form. For example Aberdeen council stipulated in its Carbon Management Programme that part of its 13 Business Travel Projects one will be responsible for the installation of electric vehicle charging points in selected Council car parks (by 2015). Or in Cambridge (CC plan 2008) the Waste and Fleet Management should trial electric powered vans and introduce recharging facilities for electric vehicles in car parks (by 2010). Another example is Exeter City Council (2008) which recognised in its climate change strategy that 21% of its carbon emissions in 2004 came from road transport and, in partnership with the Transport Authority, wants to encourage public transport providers to invest in transport fleet to deliver carbon efficiencies using e.g. hybrid models. Or Manchester City Council (2009) who stipulated that EV will be the vehicles of choice, and making highly visible charging stations available across the city.

![Figure 6-2 Climate change mitigation measures (mentioned by the 28 cities)](image)

6.3.3 EV registration and supporting public charging infrastructures

Table 6-1 shows the number of EV registered per city or region as reported by the DVLA to the House of Commons Transport Select Committee (HoC, 2012). This column however does not tell the full story. Many EV drivers lease their cars rather than buying them outright. Those vehicles are often registered by Leasing Companies which are located in London and the South East of Britain. During that reporting period for example, 44 electric vehicles were leased to the Switch EV trial in and around Newcastle upon Tyne, but were not registered in this area. For the 30 UK Urban Audit cities it was possible to use Electric Vehicle Registration data encompassing quarterly registration up to January 2015 for only 20 cities. This data is shown in the final three columns of table 2 and was used to create an additional metric representing the rate of increase in EV uptake. The argument is that if the climate change strategies are effective, then they will not only lead to an increase in the absolute number of EV, but they will also contribute to the rate of increase in uptake for EV. This data is used in Table 6-1 as a variable in the statistical tests described below.
Although less relevant in cities, the limited range of EV is still seen by many as the key barrier to the mass uptake of EV. This could be addressed in one of two ways: either the actual range of the cars needs to be improved or through an abundance of public charging infrastructure which would give drivers the confidence that they could complete their journeys and top up their charge as and when it was needed. Even though cities can address the lack of public recharging infrastructure, this has not been followed through by the cities which mentioned EV in their mitigation strategy documents as demonstrated by the analysis undertaken in this chapter. Moreover, even in cities with significant EV charging infrastructure such as Newcastle, many EV drivers still believe that more public infrastructure is needed (Hübner et al., 2014). During a EV trial conducted by our colleagues at Newcastle University, it was found that 30% of charge events took place at public charging infrastructure with 20% of EV drivers using public charging infrastructure as their primary means of charging (Hübner et al., 2014).  

<table>
<thead>
<tr>
<th>Table 6-1 UK Urban Audit cities strategies and electric cars registered\textsuperscript{13}</th>
<th></th>
<th></th>
<th></th>
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<td>2</td>
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<td>Belfast</td>
<td>N. Ireland</td>
<td>No data</td>
<td>Yes</td>
<td>No</td>
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<td>3</td>
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<td>Yes</td>
<td>Yes</td>
<td>33</td>
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<tr>
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<td>England</td>
<td>184,883</td>
<td>Yes</td>
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<td>Bristol *</td>
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<td>208,666</td>
<td>Yes</td>
<td>No</td>
<td>2</td>
<td>6</td>
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<td>Yes</td>
<td>46</td>
<td>51</td>
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<td>2,542,734</td>
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<td>Yes</td>
<td>Yes</td>
<td>56</td>
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<td>No</td>
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<td>139,717</td>
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<td>Yes</td>
<td>5</td>
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<tr>
<td>Worcester</td>
<td>England</td>
<td>50,667</td>
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<td>No</td>
<td>4</td>
<td>54</td>
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<td>Wrexham</td>
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<td>63,303</td>
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<td>No</td>
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<td>0</td>
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<tr>
<td>Total</td>
<td></td>
<td>6,177,213</td>
<td>28</td>
<td>13</td>
<td>667</td>
<td>1305</td>
</tr>
</tbody>
</table>

\textsuperscript{13} Registrations in each city based on 2011 data as reported in the evidence to the Transport Select Committee by the Driver & Vehicle Licensing Agency (DVLA) (cf HoC 2012). Cities marked (*) have additional EV quarterly registration data from 2011 to January 2015.

\textsuperscript{14} Charging in Points registered in 2014/2015
However lack of public charging infrastructure is still quoted as one of the main barriers to the uptake of electric vehicles even by those drivers who extensively used public charging facilities. This suggests that cities may have to rethink the locations they choose for EV charging points and choose highly visible and strategic locations for the placement of new charging infrastructure. Charging locations should be evaluated according to popularity and lessons need to be learned from the first Plugged-in-Places schemes which saw nearly 5,000 charging points installed across the UK (Robinson et al., 2013).

6.3.4 Statistical Test for the Effectiveness of Climate Change Strategies

The number of charging points, the proportion of EV registered and the relative change in registered EV were analysed. To understand the relative change in registered EV, a sub-set of the data was used due to the difficulty in obtaining up to date statistics on EV registration. The Shapiro-Wilk test showed that both the number of charging points and the number of EV were not normally distributed. This and the small sample size meant that non-parametric tests were used to test whether mentioning electric vehicles in their climate change strategies influenced the uptake of EV on a city-level. The Wilcoxon rank sum test was used to compare the two groups of cities. As shown in Table 6-2, there was no statistical difference between those cities who had an EV strategy and those who did not in terms of the uptake of electric cars and the number of public charging points. There is therefore no statistical difference between those cities that promote EV in their climate change mitigation strategies and those that do not. This is a worrying trend as reaching mitigation targets anticipates the uptake of electric vehicles as a new means of an urban means of transport. All those findings suggest that cities may only pay lip-service to their climate change mitigation strategies. It is therefore important that cities begin to actively encourage the uptake of EV and start to remove barriers by for example providing public charging infrastructures.

<table>
<thead>
<tr>
<th>Variable</th>
<th>W</th>
<th>p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>EV as a percentage of registered vehicles in urban audit cities</td>
<td>122</td>
<td>0.6432</td>
</tr>
<tr>
<td>Increase in registered EV (subset of 20 cities)</td>
<td>47</td>
<td>0.8534</td>
</tr>
<tr>
<td>Charging points in urban audit cities</td>
<td>82.5</td>
<td>0.2431</td>
</tr>
</tbody>
</table>

6.3.5 Projections and impacts of mitigation strategies and EV uptakes

In 2013, a total of 6,300 electric cars were registered in the UK, accounting for only 0.02% of all registered cars (Table VEH0203). While the uptake of ultra-low emission vehicles steadily increased (Table 6-3), their numbers are relatively small compared to the overall number. Even though electric vehicles are often referred to as an urban means of transport, little seems to be done on city level to encourage their uptake.

<table>
<thead>
<tr>
<th>Year</th>
<th>Plug-in-Grant Eligible Cars</th>
<th>Non Plug-in-Grant Eligible Cars</th>
<th>Plug-in Grant Eligible Vans</th>
<th>Non Plug-in Grant Eligible Vans</th>
</tr>
</thead>
<tbody>
<tr>
<td>2010</td>
<td>111</td>
<td>141</td>
<td>0</td>
<td>219</td>
</tr>
<tr>
<td>2011</td>
<td>1,051</td>
<td>153</td>
<td>34</td>
<td>221</td>
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<td>2012</td>
<td>2,198</td>
<td>132</td>
<td>264</td>
<td>159</td>
</tr>
<tr>
<td>2013</td>
<td>3,445</td>
<td>28</td>
<td>187</td>
<td>113</td>
</tr>
<tr>
<td>2014</td>
<td>14,493</td>
<td>81</td>
<td>660</td>
<td>134</td>
</tr>
</tbody>
</table>

15 Department for Transport, Table VEH0170.
Although it was estimated that up to 60% of newly registered cars in the UK could be EV by 2030 (CCC, 2015a; CCC, 2015b), for the purpose of this research, we estimate that EV will reach some 7% by 2020 and 18% of newly registered cars by 2030. Putting this into context, EV and ultra-low carbon vehicles accounted for about 0.81% of new car sales in 2014 in the UK. Nevertheless we assume here that projected demands for EV are met.

We evaluated current carbon output and how this output could change under different scenarios. It was assumed that the current vehicle fleet would be steadily replaced with new vehicles, of which a certain proportion is electric. As the new vehicles are introduced, the carbon output of the entire fleet would reduce as the older, less efficient vehicles are replaced by either more modern, carbon friendly ICE vehicles or by EV. It should be noted that the older vehicles are not necessarily being directly replaced by new vehicles. In reality new vehicles would transition into second-hand vehicles, of which a proportion would be scrapped. However, the net result is that a certain number of new vehicles will be added to the fleet whilst a certain number of old vehicles will be scrapped. In addition, the gCO2/km metric for each new ICE vehicle would improve with each year from 140gCO2/km in 2012 to a minimum of 95 gCO2/km in 2020. It is likely that there will be further efficiency savings beyond this point, but none are currently legislated for. In a similar way, it is likely that there will be an increase in the efficiency of the EV over the 85 gCO2/km value used here. Secondly, there will be improvements in the carbon efficiency of the power which is being used to recharge the EV. Values for the level of decarbonisation vary (and are strongly dependent on technological advances and future government policy) but estimates range between 30% and 50% decarbonisation by 2020 on 2011-2013 levels (CCC, 2015a). Finally, in addition to the “churn” of vehicles, there will be a steady increase in the number of vehicles on the road as dictated by the results from the Department for Transport’s National Transport Model (DfT, 2013).

![Graph showing level of carbon reduction within the private vehicle sector](image)

**Figure 6-3 Level of carbon reduction within the private vehicle sector**

Figure 6-3 shows the scale of change that is needed if the Urban Audit Cities we surveyed are meeting carbon reduction commitments. Under a 50% reduction on the 1990 levels, it can be seen that with no wide scale penetration of EV, the target will be missed by approximately 1MT of CO2. Whilst the majority of carbon savings needed can be delivered simply through the replacement of old vehicles by more efficiency new vehicles, but by itself this is not enough. However, with a 30% penetration rate coupled with a 30% decarbonisation rate it will be possible to reduce the transport emissions to the
required levels. It would be possible to extend this analysis further into 2050 but the emissions of vehicles (both ICE and EV) are strongly dependent on technology, and as yet there are no reliable indications of the extent to which technology will deliver further cost and emission reductions.

6.4 Discussions of cities strategies and EV incentives

In this chapter we argued if there is an EV specific implementation plan within the climate change mitigation strategy of a city, then it could only be judged successful if it leads to an increase in the number (or utility) of EV within that city. Unfortunately, separating the exact causes behind any particular variation in EV numbers within a city would be almost impossible due to the number of contributing factors. However, by looking at the effect of EV strategies en-mass, it is possible to judge if the use of EV specific strategies have led to an increase in EV usage. From the data shown here, none of the three main indicators of EV usage (EV as percentage of registered vehicles, rate of increase in EV usage, number of EV charge points) show any reliable statistical relationship with the presence of a specific EV policies by the cities investigated. This leads to two possible reasons:

1. Either incentives for EV purchase and use are beyond the abilities of cites to change, or
2. The climate strategies by the cities are ineffectual.

Aspects for the first point will be true for all cities. For example, in multiple surveys the price and range of EV has been brought up as a limiting factor in the purchase of such a vehicle (Hübner et al., 2012). These are factors which an individual city (beyond offering a subsidised purchase) is not able to alter. If the limiting factors for EV purchase are all on the national scale then it would be justified if cities did not include specific policies targeting EV. The second possible reason is more difficult to quantify. Whilst it may be possible to assign a cost for the implementation of any given strategy, its effectiveness is more difficult to determine. Untangling the web of behavioural influences (Harvey et al., 2014), financial decisions and unconscious biases mean that finding the “levers” that cities can pull and their effect on the populace is a complicated task. One possibility would be to further split each cities strategy into its constituent parts and the separate strategies into a series of specific actions that were taken. If an action was taken by a city (rather than simply included in the strategy) then there should be a corresponding expected result, such as free parking and charging, EV access to ‘no car lanes’ and other incentives.

Any strategy that is not resulting into action or a result that cannot be measured would be flagged up as a non-workable strategy. From our research it should be possible to build up a picture of how individual actions taken by cities affect aspects of EV uptake. The inability of cities to affect the uptake of EV is only an issue if we require the large scale adoption of EV to meet the carbon targets. However, from Figure 6-3 it can be seen that the vast majority of CO₂ reduction in the transport sector would be delivered simply by the replacement of old ICE vehicles with new, more efficient vehicles. Assuming that the current technological trend in CO₂ reduction for vehicles will continue is not an unreasonable assumption, but it is also a wager on future technological improvements which could lead to a national transport policy based on vehicle efficiencies which never arrive. However, there will be a point where efficiency improvements in ICE vehicles will become more difficult to achieve. At this point it should be possible to continue the decrease in the transport sector CO₂ emissions through the adoptions of EV, but for this strong national policies and city strategies are needed.

6.5 Conclusions and implications to meet carbon reduction targets

We investigated how the climate change mitigation strategies published by 30 UK Urban Audit cities influenced the uptake of EV and the future prospects for affecting the vehicle fleet. We have shown that strategies that included EV had no statistically significant impact on the actual uptake of EV or the introduction of public charging infrastructure. Our findings suggest that cities may pay lip-service mentioning EV in their strategies. In the short term this is not necessarily a problem as the move towards mitigation targets can be driven by the purchase of new more efficient ICE vehicles to replace older,
more carbon intensive vehicles. However there is also a trend towards targeted measures to encourage EV ownerships for fleets and private individuals as studies are now reporting on what are the more effective measures in cities to foster uptake.

The lag between national policy, city strategies and local uptake (plus the relatively long time scales for vehicle scrappage) means that for cities to meet future targets it is important to lay the groundwork now for a truly carbon friendly transport fleet. Cities must begin to actively encourage the uptake of EV, to improve the infrastructure required for the ergonomic use of EV and to remove (or at least reduce) some of the factors preventing drivers from purchasing electric cars. The failure of current strategies to increase EV uptake means that cities must learn what strategies have been successful in increasing EV uptake, whether this is from individual aspects of the strategy already used or from city wide strategies enacted both within the UK and further afield. In addition it must be considered whether there are aspects of EV uptake that are out of control of cities, e.g. consumer driven adoption of EV driven by either technological misgivings or cost considerations. For example, one apparently successful strategy has been to invest in a public charging infrastructure which is highly visible easily accessible for drivers. Yet, many cities do not seem to actively invest in public EV charging infrastructure despite their stated aims of supporting EV uptake as part of their climate change mitigation plans. Some exceptions are Birmingham, London and Newcastle upon Tyne, which are cities that have been at the forefront of introducing incentives, such as free parking, public charging infrastructure and have seen a subsequent above average uptake of EV.

We also found whilst the addition of EV into the vehicle fleet is one of many options to achieve the 2027 target, it would also be possible to achieve this with the modest assumption of a further 10gCO₂/km efficiency improvement in ICE vehicles from 2020 to 2027. The findings presented here have implications to future climate change strategies across European cities, as it has highlighted the importance of incentives such as infrastructure provisions and clear signalling by authorities. This has relevance to the ongoing RAMSES research on identifying contributing factors of transition (WP8); and stakeholder dialogues (WP9). For wider policy implications, which is part of RAMSES Task 7.3 we have shown that whilst improved efficiency in new build ICE vehicles coupled with the year on year churn of vehicles could lead to a large reduction in CO₂ this will not be enough to achieve the 2027 target and alternative modes of transport such as EV but also Biofuel needs to be considered by Central and Local Government.
7 The trinity of scaling to assess cities

Cities across the world aim to become more sustainable and RAMSES cities are no exception (Domingos et al., 2015; Mendizabal et al., 2016) and reduce vulnerability and increase efficiencies. As highlighted by RAMSES cities are both drivers of climate change and foci of climate impacts (Kropp, 2013; Pryck et al., 2014) cf. also Chapters 2, 3 and 6. In addition RAMSES D7.2 (Domingos et al., 2015), D8.1 (Mendizabal et al., 2016) and D9.1 (Terenzi and Wigström, 2014) have highlighted that cities are aware of the opportunity this provides when leveraging concentrations of activity to reduce climate risks. GHG emissions and address sustainability challenges similar to what the RAMSES project has proposed (Kropp, 2013).

In Chapter 3 we summarised and applied some of the state-of-the-art methods to benchmark the performances of cities, but what type of city is more successful in terms of reducing CO₂ emissions? In this chapter, which will be published in the peer-reviewed journal “Environment Planning B - Planning and Design” (Rybski et al., In Press), we provide quantitative methods to analyse the efficiency of city structures. Our focus is on the potential to reduce CO₂ analyse the scaling with population size (as one strong infrastructure indicator). We find that the exponent is development dependent with a transition from super- to sub-linear scaling. From the climate change mitigation point of view, the results suggest that urbanisation is desirable in developed countries. Further, we compare this analysis with a second scaling relation, namely the fundamental allometry between city population and area, and propose that density might be the decisive quantity. Last, we derive the theoretical country-wide urban emissions by integration and obtain a dependence on the size of the largest city. The results and insights obtained from this section can serve as valuable input for the understanding of urban transitions (WP8).

7.1 Introduction to scaling and city assessments

Already in the first half of the 20th century G.K. Zipf thought about how urban indicators scale with the city size and finds e.g. proportionality between the number of personal & business service establishments and the population (Zipf, 1949). More than half a century later, inspired by the analogy of the city as a metabolism, Bettencourt et al. (2007) analyse various indicators and measured super-linear scaling for quantities related to infrastructure, innovation or wealth, linear scaling for quantities of individual human needs, and sub-linear scaling for material quantities and infrastructure. Exploring these results, we present a city model (Rybski et al., In Press) based on the concept of carrying capacity that leads to alternating phases of critical growth and collapse (Bettencourt et al., 2007). Urban scaling has been investigated for several other city properties, including urban green space (Fuller and Gaston, 2009); prosocial behaviour (Arbesman and Christakis, 2011); homicides (Gomez-Lievano et al., 2012); child labour, elderly population, literacy, unemployment (Alves et al., 2013); bank card transactions (Sobolevsky et al., 2014); and agriculture (D’Autilia and D’Ambrosi, 2015). Alves et al. (2015) employ a scale-adjusted metric that takes into account allometry. The super-linear scaling, particularly of economic quantities, however, remains a mostly unsolved riddle and seems to have the character of a phenomenological law.

Different models based on human interactions and social networks have been proposed (Arbesman et al., 2009; Schlüpf et al., 2014; Yakubo et al., 2014), exploring a surplus stemming from the social network, i.e. more intense or frequent social interactions in large cities. Despite a growing body of literature addressing the origins of non-linear urban scaling e.g. (Bettencourt and West, 2010; Bettencourt, 2013), little attention has been given to the implications of accelerated socio-economic activity in cities. In a more general sense, Bettencourt and West (2010) p. 912 stated: “The many problems associated with urban growth and global sustainability, however, are typically treated as independent issues”. Especially in the global context, the question of carbon dioxide (CO₂) emissions from cities is of interest (Lankao et al., 2008; Satterthwaite, 2008; Dodman, 2009a; Kennedy et al., 2009a; Glaeser and Kahn, 2010; Sovacool and Brown, 2010; Makido et al., 2012; Tollefon, 2012; Bereitschaft and Debbage, 2013; Marconcillio et al., 2013; Minx et al., 2013; Ou et al., 2013; Jones and Kammen, 2014; Baur et al., 2015; Creutzig et al., 2015). But to what extent is urbanisation driving or mitigating climate change?
The question of whether small or large cities are more efficient in terms of per capita CO$_2$ emissions has been addressed by a few publications. Fragkias et al. (2013) found that CO$_2$ emissions of U.S. metropolitan areas scale proportionally with urban population size. In contrast, Oliveira et al. (2014) report strong super-linear scaling for U.S. cities. The authors attribute this discrepancy to the differing underlying spatial units, i.e. linearity is recovered for Metropolitan Statistical Areas (MSA) and super-linear scaling is obtained for cities defined as connected urban space (Oliveira et al., 2014). Arcaute et al. (2015) define cities in England and Wales using commuting to work and population density thresholds in order to study a large set of urban indicators. The authors report that most urban indicators scale linearly with city size and conclude that population size alone does not provide sufficient information (see also Cottineau et al. (2015)). Another perspective on various urban indicators is given by Louf and Barthelemy (2014a) who propose a model of urban growth and investigate the role of congestion (Louf and Barthelemy, 2014a; Louf and Barthelemy, 2014b). Theoretically and empirically, the authors obtain super-linear scaling for transport-related CO$_2$ emissions. Based on street network analysis, Mohajeri et al. (2015) also report super-linear scaling of transport CO$_2$ emissions with city population in Great Britain.

The research reported in this chapter follows a complementary approach. As a basic meta-study, compiled from various literature sources, it has the advantage of exhibiting better global coverage than previous studies. Grouping the cities according to the Gross Domestic Product (GDP) per capita of the corresponding countries, the results indicate non-universality since the scaling of CO$_2$ emissions versus population size depends on the economic development. While cities in economically less developed countries seem to exhibit super-linear scaling, cities in developed countries exhibit linearity or sub-linearity in respect to CO$_2$ emissions. Interestingly, another scaling, namely between population and area, resembles a similar picture when plotted as a function of time. Combining both scaling relations suggests that city density might play a decisive role. Last we discuss theoretically how the total urban emissions of a country relate to the scaling of CO$_2$ emissions versus population size, and show that the population size of the largest city non-trivially influences the country emissions. Empirically, we find that the population size of the largest city also follows a scaling relation with the population of the corresponding countries.

### 7.2 Urban CO$_2$ emission and data collection

CO$_2$ emission figures were retrieved from collections of urban GHG emission estimates published in peer reviewed journals or reports from research institutes and non-governmental organisations. We have converted all figures to CO$_2$ equivalent and use the notation CO$_2$ for simplicity. Moreover, all values refer to annual figures, again we omit "annual" for better readability. The resulting set includes information on per capita urban GHG emission for 256 cities. Published figures were largely drawn from primary sources such as city assessments covering emissions in their territories. Figures were traced back to their original publication source for verification when necessary. A range of publications account for 246 of the figures which represents about 96% of the cities (Dore et al., 2006; Brown et al., 2008; Carbon Trust, 2008; Brown et al., 2009; ICLEI, 2009; Kennedy et al., 2009b; Sovacool and Brown, 2010; Hoornweg et al., 2011).

The considered per capita emissions range from 0.1 t/cap for Rajshahi, Bangladesh (ICLEI, 2009) to 43.7 t/cap for Aberystwyth, UK (derived from (Dore et al., 2006); Carbon Trust, 2008). For 23 cities, both city and metropolitan region figures were considered, as in general they are not identical (Hoornweg et al., 2011). Since this is a basic meta-study, which relies on previously published work, we cannot exclude that incoherent GHG methodologies, sources, sectors, accounting, and spatial units have been used. In general, city greenhouse gas inventories can be affected by inconsistencies leading to uncertainties (Wattenbach et al., 2015). By pooling the data, differences should average out, assuming the methods are evenly distributed among the pooled numbers. When total emissions were not available, per cap emissions were multiplied by population figures from various sources such as United Nations World Urbanisation Prospects, EUROSTAT, citypopulation.de, etc.
7.3 City efficiency and emissions

Figure 7-1 City efficiency in terms of CO$_2$ emissions$^{16}$

$^{16}$ Per capita (annual) CO$_2$ emissions divided by corresponding country emissions versus population and regressions. The emission values have been normalised by dividing by the country emissions per capita. City emission figures have been sorted and separated into groups according to the GDP/cap of the countries. (a) Group 1 (R2=0.65): Philippines (1), Bangladesh (4), Nepal (3). (b) Group 2 (R2=0.78): China (5), South Africa (1), Thailand (2), Greece (1), Sri Lanka (4), Slovenia (1), Czech Republic (1), Portugal (1), Spain (2), Mexico (1), South Korea (2), Indonesia (1), Brazil (3), India (42), Bhutan (2) and India (42, brown open triangles). (c) Group 3 (R2=0.79): Germany (4), Italy (4), Singapore (1), Belgium (1), Finland (1), Japan (1), France (1), Sweden (1). (d) Group 4 (R2=0.75): UK (35), Netherlands (1), Canada (4), Australia (1), Switzerland (1), Norway (1) and UK (35, grey open squares). (e) Group 5 (R2=0.92): USA (122). UK and India appear twice, as part of a group and as individual country.
We begin by grouping our database of 256 urban emission figures according to the country’s GDP/cap, i.e. a balanced number of cases in each of five groups (all considered cities of a country are in the same group). Based on these groups, we analyse the values by plotting the amount of CO\textsubscript{2} emissions $e$ against the corresponding city population $s$. Dividing by the per capita emissions of the corresponding countries we make the city emissions of different countries comparable. This step is only done for normalisation reasons and would not be necessary if we had sufficient values for the various countries.

We see correlations that approximately follow power-laws with exponent $\gamma$ in Figure 7-1 and the GDP data is based on The World Bank (2013). For the linear case ($\gamma=1$) the proportionality constant is identical to the per capita emissions. The exponent $\gamma$ determines how strongly the emissions increase with city size:

- $\gamma=1$ (linear) the larger the population, the higher the emissions;
- $\gamma<1$ (sub-linear) large cities emit less CO\textsubscript{2} per capita compared to small cities; and
- $\gamma>1$ (super-linear) large cities emit more CO\textsubscript{2} per capita compared to small ones.

In addition to the evaluation by the five country groups, we also estimated $\gamma$ for the three countries with more than 10 cities in our database: India (part of group 2), UK (part of group 4) and USA (constituting all of group 5). Interestingly, our findings for the USA are consistent with the results reported in Fragkias \textit{et al.} (2013), where a $\gamma$ slightly below 1 is found.

\begin{figure}[h]
\centering
\includegraphics[width=\textwidth]{figure7-2.png}
\caption{Economic development and city efficiency in terms of CO\textsubscript{2} emissions\textsuperscript{17}}
\end{figure}

\textsuperscript{17} Determined exponents $\gamma$ versus the corresponding GDP/cap. The obtained power-law exponents $\gamma$ are plotted against the weighted average (according to the number of cities, see text) of the country’s GDP/cap for the same groups and countries as in Figure 7-1. The error-bars indicate the standard errors from fitting.
The obtained exponents $\gamma$ for each group or country are plotted against the corresponding GDP/cap values in Figure 7-2. For groups with more than one country a weighted country GDP was calculated based on the number of cities in each country in the group. The Figure suggests that countries with lower GDP/cap values tend to exhibit $\gamma>1$ and countries with higher GDP/cap values tend to exhibit smaller $\gamma$, below or close to 1. Notwithstanding the large error-bars, the results indicate that there is a difference in scaling between developing and developed countries, i.e. cities in (economically) developing countries tend to a super-linear relation between population and CO$_2$ emissions ($\gamma>1$), while cities of (economically) developed countries tend to a sub-linear relation ($\gamma<1$). This suggests a transition at $\gamma \approx 1$.

In other words, large cities seem to be more CO$_2$ efficient in economically stronger countries and more inefficient in economically weaker countries.

![Figure 7-3 Exponents of the power-law relating population and area from meta-study.](image)

While the exponent $\gamma$ describes the city emissions in terms of extensive quantities (city-wide totals), it has been argued that the intensive quantities (per capita rates) are a better representation (Shalizi, 2011). Thus, next we discuss the dependence of per capita emissions on the density of cities (Um et al., 2009). Therefore, we introduce the fundamental allometry (population area allometry, see (Stewart, 1947; Batty and Ferguson, 2011; Fluschnik et al., 2014) and references therein) relating the population and the area by an exponent $\delta$. Introducing the density $r=sl/a$, we find proportionalities and obtain that in general both quantities, density and population size, have an influence. Only for either $\delta=\gamma$ or $\delta=1$ there is a pure dependency on the density or size, respectively. The latter recovers a power-law with exponent $\gamma$.

In Figure 7-3 (Batty and Ferguson, 2011) we display $\delta$ exponents, the exponent of the relation between city area and population size, for the USA and UK as collected by Batty and Ferguson (2011). As can be seen, the reported $\delta$-values decrease along time. Qualitatively, this picture is congruent with Figure 7-2, suggesting that as countries develop, large cities become less dense and more efficient in terms of emissions. Assuming monotonous development of the GDP/cap along the years, the cross-sectional Figure 7-2 and the temporal Figure 7-3 suggest similarity of the exponents emphasising the importance of the density. This means the higher CO$_2$ efficiency of large cities in developed countries could come along with lower densities of those cities.

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$^{18}$ The 2005 value of the US is based on MSA. The exponent $\delta$ is plotted as a function of the year for the USA and the UK. In the recent years exponents $\delta<1$ are reported, indicating that large cities have smaller density than small ones.
It needs to be mentioned that scaling relations strongly depend on the definition of cities (Louf and Barthelemy, 2014b; Arcaute et al., 2015). Functional or morphological units can influence the resulting exponents, so we cannot exclude a bias if developing countries systematically use different definitions than the developed ones. A similar argument could also apply for Figure 7-3. At least the most recent δ value for the USA corresponds to MSA, which are based on a functional definition – previous measures are based on urban areas. Since functional definitions became more and more widely used after the 60s, this could also affect the trend observed for the UK.

7.4 Theoretical country-wide urban emissions and correlations

Relating emissions from cities to the emissions from an entire country could help to understand the drivers of emissions. Estimating the emissions of an entire country with the aforementioned relationship between city size and emissions requires taking into account how many cities exist for each size. The size distribution of cities is described by the Zipf-Auerbach law (Auerbach, 1913; Zipf, 1949; Rybski, 2013), which expresses that city sizes follow broad distributions. For cities it has been found that the city size distribution follows the Zipf-Auerbach with exponent ζ≈2, see e.g. Rozenfeld et al. (2011) and references therein. In addition to the assumptions about the size distribution of cities we disregard emissions from non-urban regions.

Under these circumstances, the total urban emissions of a country are calculated by the sum of the number of cities of certain population size times the typical emissions of the corresponding city size. We derive $E$, the expression for the urban emissions from the considered country for $\gamma=2$. Accordingly, for $\gamma\neq1$ a factor $f$, depending on the population size of the largest city $s_{max}$, comes into play capturing the deviations from the country population proportionality.

\begin{itemize}
  \item $\gamma>1$: The factor $f$ increases monotonously. Larger $s_{max}$ imply relatively larger country emissions.
  \item $\gamma=1$: $E$ is independent of $s_{max}$ and it is proportional to the population of the country, i.e. $d=E/S$ are the per capita emissions.
  \item $\gamma<1$: The factor $f$ decreases monotonously. As the largest city population increases, the total emissions $E$ decrease relatively.
\end{itemize}

\underline{Figure 7-4 Illustration of theoretical country-wide urban emissions$^{19}$.

We plot the rescaled emissions $E/(dS)$ (a) as a function of $s_{max}$ and (b) as a function of $\gamma$.}

---

$^{19}$ We plot the rescaled emissions $E/(dS)$ (a) as a function of $s_{max}$ and (b) as a function of $\gamma$.  

- 72 -
As indicated above, the exponent $\gamma$ reflects the development of the country, and the special case $\gamma=1$ marks a transition made by economically emerging countries. In any case, the proportionality constant $d$ and the population of the entire country $S$ remain. The normalisation done in Section 7.3 (City efficiency and emissions) levels out the allometric relations to $d=1$. However, so far we have not considered any relation between $S$ and $s_{\text{max}}$. By doing so, the transition point, based on the country emissions can be further refined. In the following we depict a third scaling relation, namely between the total population of a country and the population of the largest city.

![Figure 7-5 Correlations of country population and population of its largest city](image)

In Figure 7-5 (The World Bank, 2012) we find power-law correlations between the country populations, $S$, and the corresponding largest city populations, $s_{\text{max}}$, according to a power-law with exponent $\tau$. A similar relation has been studied between urban population of the country and population of metropolis (Pumain & Moriconi-Ebrard, 1997) – the reported exponent $1/0.81=1.23$ is close to our value. This relation can be used to eliminate the country population, $S$, and the transition between super- and sub-linear scaling with $s_{\text{max}}$ is then given by $\tau+\gamma=2$, i.e. at $\gamma<0.8$ a change occurs from super-linear (for $\gamma>0.8$) to sub-linear (for $\gamma<0.8$) influence of the largest city (neglecting the case $d\neq1$). We now use the emissions versus density curve by Newman and Kenworthy (1989) to have a glimpse on the influence of density.

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20 (a) Country population $S$ versus largest city population $s_{\text{max}}$ for 205 countries in the year 2009. Each circle represents one country, the solid blue line is a regression, and the dotted orange line is a guide for the eye with slope 1. (b) The exponent $\tau$ was obtained from regressions as in panel (a) for various years (full squares). Open diamonds represent the $\tau$ values obtained from fitting the inverse function of (Fluschnik et al., 2014). The error-bars indicate the standard errors from fitting. The colours distinguish the number of available countries: green, 204; blue, 205; and grey approximately 120 (mostly large countries). The dotted orange line indicates $\tau=1$. The exponent $\tau$ is clearly above 1, i.e. $\tau=1.2$, and minimally decreases over time, whereas the trend is smaller than the statistical error. The exponent $\tau$ is clearly above 1, i.e. $\tau=1.2$, and minimally decreases over time, whereas the trend is smaller than the statistical error.
In Figure 7-6 the values of annual gasoline use per capita as a function of urban density as extracted from Newman & Kenworthy (1989) and are plotted in double-logarithmic scale. While the authors report an exponential relationship, we fit a power-law and obtain a slope 0.92±0.07 (disregarding the point for Moscow). This result might be superposed by a size-effect as described in Section 7.3.

7.5 Discussion and conclusions of using scaling

In summary, in this chapter we elaborate on the trinity of scaling relations of urban CO2 emissions, consisting of

(i) Urban CO2 emissions vs. city population size (exponent $\gamma$),
(ii) City population vs. city area (fundamental allometry, exponent $\delta$), and
(iii) Population of largest city vs. population of country (largest city allometry, exponent $\tau$).

The analysis, based on our literature survey, indicates that $\gamma$ is non-universal and dependent on the GDP/cap of the corresponding country. A transition occurs at approximately 10,000 US$, where the emissions exponent changes from $\alpha>1$ (for small GDP/cap) to $\alpha\leq1$ (for large GDP/cap). This implies that while in developing countries small cities are more CO2 efficient, in developed countries this is the case for the large ones. Thus, we cannot make any general statement regarding whether small or large cities perform favourably in relation to CO2 efficiency.

In per capita terms, the GDP-dependence resembles an inverted U-shape, which is also known as Environmental Kuznets Curve, see (Kornhuber et al., Draft) and reference therein. It can be related to the interplay between sectoral composition and urbanisation (Lutz et al., 2013). While developing countries mostly exhibit agrarian and increasingly industrial economies with relative small but increasing fraction of urban population, developed countries exhibit service and decreasing industrial sector economies with large and consolidated fraction of urban population. Our findings can also be related to (Sarkar et al., 2015), where an income dependence of allometry is found for Australia, i.e. lowest incomes grow just linearly or sublinearly and highest incomes grow superlinearly with population. From simple considerations we obtain that, in general, the urban emissions depend on both

21 The power-law fit – disregarding Moscow (open circle) – leads to the slope 0.92±0.07. In contrast to Figure 7-1 in Newman and Kenworth (1989) the data is plotted in double-logarithmic scale based on the data given for 1980.
population size and density. The latter is characterised by the fundamental allometry (ii). Studying the corresponding exponents from Batty & Ferguson (2011) we find that δ seems to be time-dependent. Considering that the GDP/cap increases continuously over the years, we draw the analogy between the relations (i) and (ii) and hypothesise that gains of efficiency might be related to the density of the cities. Unfortunately, from our literature survey only an insufficient number of density figures could be obtained.

Nevertheless, density scaling relations for urban CO₂ emissions are analysed in Chapter 8 (City characteristics and CO₂ efficiency) and Gudipudi et al. (2016). Regarding transport emissions, Newman and Kenworthy (1989) report an exponential decrease of gasoline use per capita with increasing urban density. We displayed the data in double-logarithmic scale and fit a power-law. This is contradicting our findings and the solution might lay in the superposition of size and density scaling, as derived in Section 7.3. Another reason could be a bias in the definition of the cities investigated. Moreover, Louf and Barthelemy (2014a) derive that the emissions per capita are not a simple function of density, but rather of surface area.

With the aim of characterising country-wide urban emissions, we theoretically integrate the city emissions (assuming Zipf-Auerbach law with ζ≃2) and find that for the general case γ≠1 the population size of the largest city in a country plays an important role. Depending on the value of γ it leads to relatively increased (γ>1) or reduced (γ<1) country-wide urban emissions. Interestingly, according to the scaling relation (iii) with τ≃1.2 the above mentioned transition point can be readjusted to γ≃0.8 when considering the country-wide emissions as a function of the size of the largest city are considered. To our knowledge the scaling relation (iii) is previously unknown.

Coming back to the initially posed question about the sustainability of cities, we first need to confine the term “sustainability” to the climate change context and CO₂ emissions. From our analysis we can only give a differentiated answer, namely that urbanisation may drive climate change in developing countries and may mitigate climate change in developed ones. Further studies, measuring the exponent for each country rather than aggregating cities from various countries, are needed to confirm these results. However, our findings suggest that density-scaling might as well play a role, and we therefore call for a coherent analytical and empirical analysis that takes into account both the population size and the population density and investigate this further in Chapter 8. These findings are of relevance to the wider research by the RAMSES consortium, and especially on the transition research in WP8.

In addition, we would like to emphasise that data on city emissions is still insufficient and we urge for a systematic and comparable reporting of CO₂ emissions from cities, for example as recently proposed (C40 ICLEI, 2012). The same holds for alternative data sources, such as data from the Greenhouse gases Observing SATellite (GOSAT) (Hammerling et al., 2012) and local detailed GHG monitoring programmes, e.g. (Tollefson, 2012).
8 City characteristics and CO₂ efficiency

One important aspects of RAMSES is to identify urban characteristics and their inter-linkages to urban mitigation and adaptation efforts and described in the DoW T1.3 researches quantitative methods to analyse different levels of city efficiencies (Kropp, 2013). It is expected that such an in-depth understanding of local features will enable local decision makers to find appropriate strategies (cf. Chapters 2 and 5) to increase mitigation and adaptation efforts at local scale. As we discussed in Section 6.4 certain initiatives aiming to mitigate GHG emissions such as introducing electric vehicles might be well beyond the scope of local governments. However, urban population density, as mentioned in Chapter 7, plays a decisive role in GHG emissions globally.

We applied benchmarking methods to assess whole cities in Chapter 3 and utilised quantitative methods to analyse the efficiency of individual cities characterises in Chapter 7. In this chapter, which is based on the paper that will be published in April 2016 in the peer-reviewed journal “Energy Policy” (Gudipudi et al., 2016), we describe the relationship between population density and the CO₂ emissions. We do this for some of the RAMSES cities but also apply this for other relevant cities. As the physical typology of buildings and infrastructures has been described by RAMSES D2.2 (Acero et al., 2014) and urban population density is relatively well within the control of local governments cf. Chapter 2 we limit this chapter to the analysis of population density and efficiencies. It could be clearly argued that the population density is probably the most prominent feature that differentiates cities from their rural counterparts.

8.1 Introduction for assessing city efficiencies

While rapid urbanization has accelerated innovation and socio-economic growth on one hand, it has contributed to generate a multitude of global problems on the other hand ranging from climate change and its impacts on food, energy, water availability, public health and global economy (Bettencourt and West, 2010). Consequently, sustainable management of urban areas worldwide is considered as one of the main challenges of the 21st century. Cities consume between 67-76% of the global energy supply and release 71-76% of the carbon dioxide emissions (Seto et al., 2014). A major share of which can be attributed to the building and transportation sectors, thus specific infrastructural features of a city. Finding effective and integrated land use and transportation solutions to decrease GHG emissions are key challenges.

However, tackling such challenges need a detailed and systematic analysis (Domingos et al., 2015; Mendizabal et al., 2016) which is constrained by major and prominent problems of definitions that was raised in Chapter 1 and throughout this report. For example there is no universal definition of what qualifies a given settlement as urban or rural (UN, 2014). Environmental assessments become even more problematic when one needs to distinguish between administrative entities and/or suburbs, or to analyse twin cities. Thus, it is also difficult to relate carbon footprint of a city to its physical features and to derive conclusions about how urban structure needs to be designed to become more sustainable (Bramald et al., 2015; Vemury et al., 2015). For example, Parshall (2010) highlighted that the methodological challenges in modelling energy consumption and CO₂ emissions at urban scale are extremely problematic, because of a lack of a standard definition of what is urban and rural. However, there exists a general consensus that population density in settlements influences the on-road energy consumption and building emissions (Newman and Kenworthy, 1989; Dodman, 2009b; Boyko and Cooper, 2011; Jones and Kammen, 2014; Lee and Lee, 2014). Therefore, amongst others, population density is considered to provide a suitable indicator to measure the relationship between certain urban “functions” (for e.g. housing, traffic, recreational and institutional facilities) and their sustainability. Since urbanization is an inevitable global phenomena; it stresses the importance of assessing the potential influence of population density in decreasing carbon footprint and improving energy efficiency. Previous studies focusing on population density and their energy consumption and/or GHG emissions in cities can essentially be broadly categorized into 3 groups:
1. Studies focusing on the relationship between population density and transport related energy consumption (Newman and Kenworthy, 1989; Brehey and New, 1995; Kenworthy et al., 1999; Mindali et al., 2004; Tiwari et al., 2011; Wang et al., 2014)

2. Studies focusing on the relationship between population density and household energy consumption (Lariviére and Lafrance, 1999; Ewing and Rong, 2008; Martilli, 2014; Ye et al., 2015; and

3. Cross sectoral studies focusing on population density, household carbon footprints and/or GHG emissions (VandeWeghe and Kennedy, 2008; Brown et al., 2009; Jones and Kammen, 2014; Lee and Lee, 2014; Myors, 2014).

The methodologies used in most of the previous studies to find out the relationship between population density and energy consumption or GHG emissions, though internally consistent; vary significantly. Therefore the impact of population density on energy consumption in these studies exhibited conflicting results. Using existing urban GHG emission inventories to find out this relationship is also debatable because of the differences in spatial scales (urban extents) and emission estimation i.e. accounting methods, scope of GHG’s and emission sources (Wintergreen et al., 2006; Satterthwaite, 2008; Kennedy et al., 2009a; Dhakal, 2010; Ibrahim et al., 2012; Fong, 2014).

The approach followed in this work systematically overcomes the aforementioned ambiguity in the definition of urban areas (and therefore their extents and population density) by applying a procedure called ‘City Clustering Algorithm’ (CCA) on gridded land use and population datasets. For further details see (Rozenfeld et al., 2008). The CCA considers cities as adjacent populated clusters. The algorithm is an automated and systematic way of identifying adjacent populated clusters based on geographic location of people and combining them into one city cluster. The population densities of the resultant city clusters are comparable since the rationale behind identifying each city cluster is the same.

The CCA method together with the availability of gridded sectoral emission data from the Vulcan Project (Gurney et al., 2014) paved road towards more systematic analysis on the influence of urban population density on sectoral emissions. By restricting the analysis to on-road and building (residential and commercial) sectors, the emissions were assigned to the populated settlements which are later aggregated to city clusters identified by the CCA. Consequently, our analysis clearly shows that there exists a fundamental relationship between population density and sectoral emissions on a per capita basis (which will be referred to as CO\textsubscript{2} efficiency from hereon). Our results show that doubling the population density will improve CO\textsubscript{2} efficiency in building and on-road transportation sectors at least by 42%.

8.2 Data gathering and processing

8.2.1 Data sources

The population count data used in this study is obtained from the Global rural-urban mapping project (GRUMP) for the year 2000 available at a spatial grid resolution of ~1 km\textsuperscript{2} (CIESIN, 2011). Since Potere and Schneider (2007) indicated that the spatial extents of urban or non-urban land use varied depending on the methodology used to classify them; we extended our methodology and checked the consistency of the results for two different land use/land cover data sets namely: (1) The GRUMP classification of global land use into urban or non-urban at a spatial grid resolution of ~1 km\textsuperscript{2} and (2) Global land cover 2000 (GLC) data available at ~1 km\textsuperscript{2} grid spatial resolution (Bartholome et al., 2003).

The emissions data used in this research is obtained from the Vulcan project (Gurney et al., 2014) which provides a consistent sectoral CO\textsubscript{2} emission data for entire US at a grid resolution of ~100 km\textsuperscript{2} (which will be further referred as Vulcan grid). We restricted our analysis to on-road transportation and buildings (residential and commercial) sectors. However, it has to be noted that the CO\textsubscript{2} emissions from electricity are allocated at geocoded locations i.e. at source instead of allocating them at the point of consumption in the Vulcan grid. The building emissions in the Vulcan grid therefore predominantly include the CO\textsubscript{2} emissions related to combustion of fuels for heating purpose. The on-road mobile emissions are based on county-level data from the National Mobile Inventory Model (NMIM) County
Database (NCD) for 2002 which quantifies the vehicle miles travelled in a county by month, specific to vehicle class and road type. The Mobile 6.2 combustion emissions model is used to generate CO₂ emission factors on a per mile basis given inputs such as fleet information, temperature, fuel type, and vehicle speed (Gurney et al., 2014).

8.2.2 Analytical concept and data pre-processing

The data pre-processing done in this research comprised of three major steps. As a first step in data processing, we did a spatial overlay of the gridded population data separately on the GLC and GRUMP land cover datasets. The pre-conditions for the spatial overlay are: (1) the grid cell has to be classified as ‘urban’ and ‘artificial surfaces/associated areas’ in the GRUMP and GLC land cover datasets respectively and (2) population count of the grid cell attributed to urban/artificial surfaces land use in GLC and GRUMP land cover datasets is at least 1. As a result of the first step; we attributed population count to urban/artificial land use separately for GLC and GRUMP land cover datasets at ~1 km² grid cells.

As a second step, we superimposed the sectoral emissions in the Vulcan grid on the spatial overlay obtained from the previous step. Since the emissions data is available at a different resolution (~100 km²) compared to the spatial overlay in first step (i.e. ~1km²); there is a need to harmonize the datasets and bring all the datasets to same resolution. We harmonized the spatial resolution of these two datasets by the following assumption: since the study focusses only on building and on-road emissions i.e. settlements where people reside and commute, the energy consumption and the subsequent CO₂ emissions also originate from these settlements. Therefore, if a populated settlement intersects the Vulcan grid during the spatial overlay, we assumed that all the sectoral emissions in the Vulcan grid are emitted by the intersecting populated settlements. With this assumption we resampled the sectoral emissions in ~100 km² Vulcan grid to ~1km² using nearest neighbourhood method. The pre-condition for resampling are: (1) the Vulcan grid that attributes the emissions to the populated settlements should have non-zero residential emissions; (2) emissions should be always attributed to those cells which are classified as urban/artificial surfaces land use; (3) emissions attributed to the settlements should be proportional to the area of the settlements. As a result of the second step we have the sectoral emissions attributed to the populated settlements separately for GLC and GRUMP land cover datasets.

Lastly, we applied the CCA to the both land cover datasets separately to identify the unique settlement extents (called as city clusters from hereon), their corresponding population and sectoral emissions. The CCA defines a city as a cluster of connected populated cells with a maximum size using two threshold parameters:

1. A population threshold \(P\) which defines the minimum population of the grid cell; and
2. A distance threshold \(l\) which defines the cut-off distance between two spatially contiguous cells.

A grid cell \((i)\) in the population grid is considered to be occupied if its population is more than the threshold population (i.e. \(P \geq P\)) and if it is within the distance threshold (i.e. \(l \leq l\)) to the next populated cell. In this study, we didn’t define any population threshold \((P)\), as we wanted to capture the relationship between population density and sectoral CO₂ emissions of all inhabited areas in the US. We therefore applied CCA to all grid cells with non-zero population separately to both land cover data sets from 1 to 10 km threshold distance \((l)\). The application of CCA is illustrated in Figure 8-1 where the blue colour cells are exemplified as populated cells and the empty cells as white coloured. Starting from an arbitrary populated cell which is converted to red colour as shown in Figure 8-1(a), the algorithm starts iterative burning of all the neighbouring cells that are within the given threshold distance \((l)\) into one city cluster converting them to red colour as shown in Figure 8-1(b) and Figure 8-1(c). The algorithm continues this iteration until all populated cells within the given threshold distance become one city cluster (Figure 8-1(d)). For further details on CCA, see (Rozenfeld et al., 2008).
As the sectoral emissions are always attributed to the populated settlements; the emissions of the city clusters are also aggregated proportionally. As a result of the third step, we obtained a number of city clusters (n) at varying threshold distances (1 to 10 km), their sectoral emissions, extents, population and subsequently their population density separately for GRUMP and GLC land cover datasets (which will be further referred as GRUMP and GLC data in this chapter). Since the criteria used to identify these city clusters is the same for all inhabited areas in the grid; their extents, population and subsequently their population density are also comparable unlike the traditional urban administrative boundaries. We used the population density of city clusters obtained from the CCA and their corresponding sectoral emissions to find out the influence of population density on CO$_2$ efficiency. Table 8-1 gives an overview about the datasets and methods used in our study.

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22 (a) Burning an arbitrary populated cell from blue to red; (b) Identifying the nearest populated cells within the given threshold distance (I) and converting them into red color; (c) Iterative burning of all populated cells until all cells within the threshold distance (I) are burned; (d) Resulting city cluster at a given threshold distance (I).
### Table 8-1 Methods used to assess city density and CO₂ emissions

<table>
<thead>
<tr>
<th>Data</th>
<th>Sources</th>
<th>Spatial Resolution</th>
<th>Methods</th>
<th>Remarks</th>
</tr>
</thead>
<tbody>
<tr>
<td>Population count</td>
<td>GRUMP population count in US for the year 2000: <a href="http://sedac.ciesin.columbia.edu/data/set/grump-v1-population-count">http://sedac.ciesin.columbia.edu/data/set/grump-v1-population-count</a></td>
<td>30 arc seconds (~1km²)</td>
<td>Spatial overlay to attribute population to GRUMP urban extents and GLC urban and built up cells.</td>
<td>Since urban extents vary from one land use data to another; we tested our methodology and compared the results on two different urban land use/extent data sets.</td>
</tr>
<tr>
<td>Sectoral CO₂ Emissions</td>
<td>Project Vulcan for the year 2002*: <a href="http://vulcan.project.asu.edu/research.php">http://vulcan.project.asu.edu/research.php</a></td>
<td>300 arc seconds (~100km²) resampled to 1km²</td>
<td>Emission attribution to the populated settlements using nearest neighbourhood method proportionally to their area.</td>
<td>The main assumption for emission attribution is that sectoral emissions originate from the populated areas. The resulting dataset consisted of building** (residential and commercial) and on-road emissions of all populated settlements in GRUMP and GLC urban/built up area cells.</td>
</tr>
<tr>
<td>Urban extents/land use</td>
<td>GRUMP Urban Extents: <a href="http://sedac.ciesin.columbia.edu/data/set/grump-v1-population-count">http://sedac.ciesin.columbia.edu/data/set/grump-v1-population-count</a>, GLC: <a href="http://bioval.jrc.ec.europa.eu/products/glc2000/products.php">http://bioval.jrc.ec.europa.eu/products/glc2000/products.php</a></td>
<td>30 arc seconds (~1km²)</td>
<td>Application of CCA to aggregate populated urban/built-up settlements into city clusters. The sectoral emissions are aggregated proportionally to city clusters.</td>
<td>This application ensured a consistent definition for identifying city extents. The resultant dataset consisted of city clusters from 1 to 10 km threshold distance; their extents, population density and sectoral emissions.</td>
</tr>
</tbody>
</table>

* Only CO₂ emissions (not all GHG’s) are reported in the project. ** Building emissions doesn’t include emissions from electricity consumption which are reported at point of production.

### 8.2.3 Emission distribution in the selected datasets

Since the methodologies used to classify urban and non-urban areas vary from the GRUMP to GLC land cover data, the extents of the city clusters and therefore the emissions attributed to these areas also varied. The spatial extent of urban clusters plays a crucial role in defining the population density of the city cluster and therefore its relationship to the emissions per capita. At a threshold distance of 1 km i.e. the least logical threshold distance that can be applied in CCA, the total area of city clusters in the GRUMP data is approximately 9.6 times to that of the total area of urban clusters in the GLC data. At the same threshold distance; the sum of residential, commercial and on-road emissions under the GRUMP data are 1.4 times to that of GLC dataset. Figure 8-2 summarizes the differences between the key parameters in GRUMP and GLC data at 1km threshold distance.
Since the extents of city clusters under GRUMP data are bigger when compared to that of GLC data; the total area of the city clusters (Figure 8-2(a)), the total population of city clusters (Figure 8-2(b)) and the sectoral emissions attributed to these city clusters (Figure 8-2 (c)) are also more in GRUMP data. As the threshold distance increased, the number of city clusters \((n)\) decreased twice in GRUMP data when compared to the GLC data \((n= 4585, 3285, 2786 \text{ for GLC data and } n= 5182, 2156, 1538 \text{ for GRUMP data at 1, 5 and 10 km threshold distance respectively})\). The gridded population is overlaid on the landcover maps, which varied and as a result so do the CO\(_2\) emissions. Though the total emissions remained the same as the threshold distance increased (since the emissions are always attributed to the populated cells); the minimum, mean and maximum emissions in each sector varied significantly for both the data sets. For instance; the city cluster emitting maximum total on-road emissions for GRUMP dataset at 10 km threshold distance is 2.1 times when compared to 1 km threshold distance, indicating the expected increase in on-road emissions as city clusters increase in size. With respect to GLC dataset, this number is bit lower (1.9 times). The population density also varied significantly in the GLC and GRUMP data sets. The city cluster with maximum mean population density in GLC dataset at 1 km threshold distance (7603 inhabitants per km\(^2\)) is almost double to that of GRUMP dataset (3860 inhabitants per km\(^2\)).

8.3 Data analysis and influences of population density

8.3.1 Sensitivity of the results to the datasets

We found out that irrespective of the land cover data set used; the total emissions (sum of residential, commercial and on-road emissions) always decreased with increase in cluster population density on a per capita basis. The results however altered in magnitude. We fitted a linear regression model between the total emissions per capita and the population density of all urban clusters for both datasets to find out the statistical relationship between the population density of the city clusters and their CO2 efficiency using the following model:
\[ \ln(\text{CO}_2/\text{Population}) = A + \beta \ln(\text{Population/Area}) \]

**Equation 8-1 Emissions per capita and the population density**

Where CO\(_2\) refers to the sum of residential, commercial and on-road emissions; \(A\) is a constant, \(\beta\) is the slope of the linear relationship in log-log scale (in natural logarithm); Population is the sum of population in each cell that form the city cluster and Area is the extent of each city cluster. Table 8-2 summarizes the results of the linear fitting at 1, 5 and 10 km cluster distance. Though there is no drastic change in the coefficient of determination (R\(^2\)) with increase in threshold distance, the slope (\(\beta\)) became steeper for both GLC and GRUMP datasets.

**Table 8-2 Linear fitting of cluster distance**

<table>
<thead>
<tr>
<th></th>
<th>1km</th>
<th>5km</th>
<th>10km</th>
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<tbody>
<tr>
<td></td>
<td>A</td>
<td>(\beta)</td>
<td>R(^2)</td>
</tr>
<tr>
<td>GLC</td>
<td>6.89</td>
<td>-0.78</td>
<td>0.69</td>
</tr>
<tr>
<td>GRUMP</td>
<td>5.17</td>
<td>-0.9</td>
<td>0.66</td>
</tr>
</tbody>
</table>

However, the slope (\(\beta\)) varied from one dataset to the other. Therefore it can be inferred that as the size of the cluster increases, the rate at which per capita emissions decrease is sensitive to the land use dataset used. Considering the least slope (\(\beta\)) from the two land use datasets used in our analysis (GLC dataset at 1km cluster distance where \(\beta=0.78\)) as shown in Figure 8-3; our results showed that doubling the population density will increase the CO\(_2\) efficiency at least by 42%. At higher threshold distances in the GRUMP data we identified that a majority of cells merging at a certain density range (4-6 in natural logarithmic scale). This phenomenon can be attributed to the proximity of the city clusters in the north and west coast considering the methodology used to prepare the GRUMP urban extents (i.e. continuous night lights). In order not to over or under estimate urban extents, we show our results only for the 5 km threshold distance. Figure 8-3 also shows the relationship between cluster density and total CO\(_2\) efficiency for GLC and GRUMP data respectively at 5 km cluster distance.

**Figure 8-3 Population density and total emissions (per capita) at 5km threshold distance**

Individual linear regression to find out the influence of population density individually on sectoral CO\(_2\) efficiency (residential, commercial and on-road emissions per capita) revealed that increasing density majorly improves the CO\(_2\) efficiency of on-road emissions compared to building (residential and commercial) sector. Our analysis (at 5 km threshold distance) showed that doubling the population density will improve the CO\(_2\) efficiency (considering the least \(\beta\) values) at least by 43%, 41% and 36% for on-road, residential and commercial emissions respectively. Table 8-3 shows how cluster population density influences sectoral emissions per capita in both the data sets at 5 km cluster distance.
### 8.3.2 Population density on CO₂ efficiency in highest CO₂ emitting city clusters

To further investigate the major emitters of CO₂ and the influence of population density on their sectoral CO₂ efficiency; we made subsets of the top 500 emitting urban clusters for both GLC and GRUMP datasets. We used the central latitude and longitude coordinates of the metropolitan statistical areas (MSA’s) as defined by the Office of Management and Budget in the US to identify the city clusters in selected metropolitan areas (U.S. Census Bureau, 2010). Figure 8-4 shows the relationship between cluster population density, buildings (sum of residential and commercial emissions) and on-road emissions per capita for the top 500 emitting urban clusters for the GLC and GRUMP data at 5 km threshold distance.

As can be seen in Figure 8-4 all cities (except San Francisco) are rated in the lower-right corner which represents that most of the highly populated cities in the US, which are also associated with higher population densities and relatively lower emissions on a per capita basis. It shouldn’t be inferred that San Francisco urban cluster has lower population density and higher emissions per capita. The reason for high emissions at lower cluster distance in San Francisco occurred because of the resampling of emissions and lower population count in the city clusters as result of lack of continuous urban land (in this case islands). At higher threshold distances, San Francisco also has higher density and lower emissions per capita. Since the electricity emissions in the Vulcan emission data are geo referenced to the production sites, cities in the south such as Los Angeles, Phoenix and Miami have relatively less building emissions (due to lower demand for heating) when compared to Chicago or New York. As mentioned earlier, population density has much higher influence on the on-road emissions than building emissions per capita for both the data sets used.

---

**Table 8-3 Cluster population and emissions**

<table>
<thead>
<tr>
<th></th>
<th>Residential Emissions per capita</th>
<th>Commercial emissions per capita</th>
<th>On-road emissions per capita</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>R²</td>
<td>β</td>
<td>R²</td>
</tr>
<tr>
<td>GLC</td>
<td>0.5</td>
<td>-0.74</td>
<td>0.43</td>
</tr>
<tr>
<td>GRUMP</td>
<td>0.36</td>
<td>-0.93</td>
<td>0.39</td>
</tr>
</tbody>
</table>
The building emissions per capita in the top 500 emitting clusters showed significant relationship to the population density in the GLC data ($\beta = -1.15$) when compared to GRUMP data ($\beta = -0.27$). This again could be attributed to the methodology used to identify urban extents in GLC and GRUMP landcover datasets. Table 8-4 shows the top 5 urban clusters ranked based on their total emissions with their corresponding sectoral emissions. Though New York urban cluster is the top emitting urban cluster in both datasets, the per capita emissions of New York urban cluster is lower when compared to other urban clusters in the colder regions (lower per capita emissions in Los Angeles urban cluster is because the relatively lower building emissions per capita). New York urban cluster emits the highest on-road emissions but again on a per capita basis, is found out to be the lowest (except for Los Angeles – Riverside cluster in GRUMP dataset). The phenomena of decreasing on-road emissions with increase in population density can be observed in the table however no such trend can be seen with respect to building emissions.
### Table 8-4 Top 5 urban clusters in terms of emissions

<table>
<thead>
<tr>
<th>City Cluster</th>
<th>Population density of the cluster</th>
<th>Sum of sectoral emissions (MT of CO₂)</th>
<th>Total CO₂ emissions (MT of CO₂ per capita)*</th>
<th>Buildings sector CO₂ emissions per capita</th>
<th>On-road CO₂ emissions per capita</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>GLC dataset</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>New York</td>
<td>2661</td>
<td>22.92</td>
<td>2.2</td>
<td>1.14</td>
<td>1.06</td>
</tr>
<tr>
<td>Los Angeles</td>
<td>2732</td>
<td>13.57</td>
<td>1.52</td>
<td>0.33</td>
<td>1.19</td>
</tr>
<tr>
<td>Chicago</td>
<td>1864</td>
<td>13.06</td>
<td>3.04</td>
<td>1.46</td>
<td>1.58</td>
</tr>
<tr>
<td>Washington (DC)</td>
<td>1516</td>
<td>9.7</td>
<td>2.81</td>
<td>1.01</td>
<td>1.8</td>
</tr>
<tr>
<td>Detroit</td>
<td>1259</td>
<td>8.1</td>
<td>2.9</td>
<td>1.16</td>
<td>1.74</td>
</tr>
<tr>
<td>New York,- Philadelphia- Washington</td>
<td>731</td>
<td>65.59</td>
<td>1.95</td>
<td>0.86</td>
<td>1.09</td>
</tr>
<tr>
<td><strong>GRUMP dataset</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Los Angeles-Riverside</td>
<td>1068</td>
<td>22.79</td>
<td>1.31</td>
<td>0.31</td>
<td>1</td>
</tr>
<tr>
<td>Chicago-Milwaukee</td>
<td>589</td>
<td>20.77</td>
<td>2.11</td>
<td>0.94</td>
<td>1.17</td>
</tr>
<tr>
<td>Boston-Worcester</td>
<td>436</td>
<td>14.58</td>
<td>2.47</td>
<td>1.18</td>
<td>1.29</td>
</tr>
<tr>
<td>Detroit-Ann Arbor</td>
<td>491</td>
<td>13.8</td>
<td>2.51</td>
<td>0.96</td>
<td>1.55</td>
</tr>
</tbody>
</table>

All emissions are in tonnes of CO₂ for the year 2002. The population density is expressed in inhabitants/km². *Sum of the total emissions (in tonnes) from the residential, commercial and on-road sectors of the city cluster divided by the population of the cluster.

### 8.4 Discussions of population density and emissions

This chapter addresses the earlier challenges in finding out the relationship between population density and GHG emissions using the CCA approach of identifying the urban extents and gridded CO₂ emission data. Through our analysis we found out that population density is one of the vital factors that influence CO₂ efficiency in cities. We showed how the emissions per capita decrease as a general trend with increase in population density. However, as showed in the previous section, the results are sensitive to the input land cover data and the threshold distance used in the CCA. The selection of the land cover data and threshold distance should be therefore done cautiously while using the CCA approach in finding this relationship.

Urban areas are often blamed as the main reasons for global anthropogenic emissions (Hoornweg et al., 2011) and are therefore identified as the key players in the global mitigation agenda (Dhakal, 2010). While it is imperative that anthropogenic emissions concentrate at areas of human activity; the natural advantage cities offer is their relatively lower emissions on a per capita basis. Per capita emissions enable intercity comparison and benchmarking of policies and infrastructure services which can be adapted to other cities. Although this study is confined to inhabited settlements in the US, the findings in the study are not unique to the cities in US itself as they broadly reflect the general consensus in urban policy discourse globally (OECD, 2012; Seto et al., 2014). However, the slope (β) of the log linear relationship between the emission efficiency and population density depends significantly on the urban form of the cities and their current building and on-road transportation infrastructure. The slope (β) might be pronounced more in case of cities in countries which exhibit more variance in population density profiles such as US when compared to cities in countries which exhibit relatively less variance in population density profiles like those in Europe and Asia. Recent studies in European and Asian cities with relatively less variance in population density profiles found out that urban morphology, local climate, household size and personal wealth influence GHG emissions more than population density (Makido et al., 2012; Baur et al., 2014) (Reckien et al., 2007). Therefore, it shouldn’t be inferred that increasing population density will inherently decrease the building and on-road emissions especially for cities in developing countries where densities are already high. Even dense settlements with poor building insulation, lack of access to public transportation facilities and on-road infrastructure might lead to further increase in GHG emissions.
Since the CCA identifies all populated cells within a given threshold distance as one city cluster, the resultant population density of the city cluster is only a weak measure of population density. Therefore the population density of Los Angeles cluster (2732 persons/km²) for example is more than the population density of the New York cluster (2661 persons/km²) since the extent of New York City cluster also includes the city cluster of Philadelphia.

8.5 Conclusions and policy implication in increasing efficiencies

In this era of rapid urbanization, resource consumption and GHG emissions associated with it; there exists a broader debate in contemporary research if large cities are efficient compared to smaller ones. However; urbanization as an inevitable phenomena, coupled with cumulative actions at local/city scale to curb GHG emissions play a significant role in global mitigation agenda. The findings of this research inherently imply that planning policies aiming at reducing urban energy consumption should contemplate on increasing population density coupled with stringent energy efficient building renovation/regulation codes, improving accessibility to public transportation and encouraging mixed land use. Recent studies (Creutzig et al., 2015) have also highlighted the significant role played by urban form in mitigating energy consumption in cities where infrastructure is still nascent. Nevertheless, a practical repercussion while implementing such smart growth policies is the immediate increase in the local land prices. Urban intensification and improved access to public transportation triggers the highly speculative local land prices.

The findings of this study suggest that one can no longer view a city in isolation with clearly defined fictitious boundaries but rather as a dynamic cluster of entities constantly evolving in space and time. Therefore urban policies aiming at curbing the energy resources consumed by these clusters should not only aim at smart growth policies within the city but also at a broader metropolitan/regional level including the satellite towns which are well beyond the current city boundaries. Nevertheless, the ultimate objective of sustainable urban management could only be accomplished through stringent enforcement strategies and effective management of urban land use as discussed in Chapters 2 and 4.
9 Main conclusions from this report (D1.3)

As described by RAMSES research, cities are places of economic activities and innovation, but also major emitters of carbon dioxide and, due to their high population and infrastructure density, sensitivity hot spots with regard to climate change impacts (RAMSES, 2014). In other words they are a large part of the global climate problem but also a nuclei for change as described in Chapter 7 (Rybski et al., In Press). The RAMSES project and T1.3 specifically has built up case studies, some of which are grouped into focal and supportive cities. The focal cities serve as exemplary sites for the development and application of tools and methods developed within the project (Kropp, 2013). The supportive cities have served for testing individual tools and made comparisons possible, specifically in the cost inventory chapter. In this report we have described existing and innovative urban benchmarking studies that we conducted for various infrastructure systems, which “on the one hand provide new insights into sustainable city characteristics and one the other hand is easily interpretable by decision-makers” (Kropp, 2013). In order to achieve the aim we have applied (and continue to apply) a variety of research approaches and findings as summarised in Table 9-1.

Table 9-1 Research summary of D1.3

<table>
<thead>
<tr>
<th>Research approach</th>
<th>RAMSES reports and summary of this research</th>
</tr>
</thead>
<tbody>
<tr>
<td>Benchmarking</td>
<td>Building on the research reported in RAMSES D7.1 (Pryck et al., 2014) we explored the relation between the centrality features, the accessibility networks and the diversity of facilities across European cities and countries and benchmarked the local efforts in terms of climate change strategies, socio-economic efficiency and environmental impacts (Chapters 2 and 3). Consulting RAMSES D2.2 (Acero et al., 2014), D2.4 (Kallaos et al., 2015a) and D5.1 (Floater et al., 2014) we gathered and benchmarked the costs of constructing or restructuring large and small infrastructure systems focusing on specific adaptation and mitigation in Chapter 4.</td>
</tr>
<tr>
<td>Case studies</td>
<td>As described by RAMSES D2.1 (Kallaos et al., 2014), D2.3 (Kallaos et al., 2015b) and D2.4 (Kallaos et al., 2015a) green infrastructure and electric vehicles are important city components and we conducted various case studies to provide quantitative assessments for such infrastructure systems. We presented two case studies on Green Urban Infrastructures and Electric Vehicle uptake and its supporting infrastructures in Chapters 5 and 6.</td>
</tr>
<tr>
<td>Generic approaches</td>
<td>Considering the research reported in RAMSES D1.2 (Boettle et al., 2016) we used generic approaches in Chapters 7 and 8. We assessed and where feasible provided quantitative methods that help analysing the efficiency of individual city structures and technologies to reduce CO₂ emissions (i.e. mitigation) or in terms of its internal resilience (i.e. adaptation).</td>
</tr>
</tbody>
</table>

As stated in Description of Work (DoW) it was envisaged that parts of our research in Work Package 1 (Kropp, 2013) should result in peer reviewed journal papers. Based on the research conducted in T1.3 a range of papers have already being accepted, published, submitted or are in draft form for submission to peer reviewed scientific journals as described in the introduction (Table 1-1). In this concluding chapter we summarise the main research areas and conclusions from each chapter, followed by a brief description of the relevance of the work to the ongoing and future research of the RAMSES consortium.

9.1 Main research areas and conclusions

In Chapter 2 we provided evidence that the existence of global policies or national legislation or similar governance structures do not guarantee the development of city plans that support climate change mitigation and adaptation efforts. Although an influence of national government frameworks on city plans was documented in some countries (Stecker et al., 2012), a national framework is not always sufficient to trigger climate change action in the city. We concluded that multiple interests and motivations are inevitable and that no archetypical way of planning for climate change exists across Europe that would help to provide the necessary infrastructures.

In Chapter 3 we benchmarked socio-economic outcomes considering the environment pollution inputs for European cities, which revealed that the efficiency of cities is independent of the size of the city. We concluded that defining efficiency of the city as the ratio of its socioeconomic outputs to its environmental inputs is feasible. But, apart from well-established and robust socio-economic background, we showed how the efficiency of cities does vary with the benchmarking methods that was applied.
In Chapter 4 we provided ballpark figures and detailed cost inventories for 9 infrastructure components, namely: air conditioning, double glazing, green roofs, levees, loft insulation, mechanical ventilation, permeable paving, solar blinds and solar panels. We concluded that more data tends to be available for countries with higher economic power. In particular, RAMSES case study cities with higher Gross Domestic Product (GDP) per capita had the largest amount of data available. We highlighted the need for transferrable methodologies that minimize the need for detailed economic data.

In Chapter 5 we demonstrated that the Performance Index (PI) can be applied in real conditions and will be a useful tool that helps city planners to select the most suitable species in order to maximize the mitigation and adaptation potential of Green Urban Infrastructure (GUI) vegetation. We concluded that developing more resilient street plantation infrastructure, specifically in the context of scattered urbanization pattern with low-density development, commonly witnessed in the peri-urban regions, will support climate change adaptation and mitigation efforts.

In Chapter 6 we argued that if specific Electric Vehicle (EV) policies or strategies for a city exist then more EV and supporting infrastructure should be present. Analysing the strategies, car registrations and the EV supporting infrastructures has shown that this is not the case for the 30 UK cities we investigated. We concluded that cities must be more proactive and understand what policies have been successful in increasing EV uptake, whether this is from individual aspects of the policies already used or from city wide policies enacted both within the UK and further afield.

In Chapter 7 we provided quantitative methods to assess efficiency of city structures and focused on the potential to reduce CO\(_2\) by analysing the scaling with population size (as one strong infrastructure indicator). Our findings do suggest that density-scaling might as well play a role, and we therefore studied the influence of population density on urban CO\(_2\) emissions in Chapter 7. We concluded that urbanisation may drive climate change in developing countries and may mitigate climate change in developed ones.

In Chapter 8 we described the relationship between population density and the CO\(_2\) emissions. We do this for some of the RAMSES cities but also apply this for other relevant cities. We showed how the emissions per capita decrease as a general trend with increase in population density. Closing the circle and coming back to the questions posed in Chapter 2 we concluded that there is a dire need for future research in this area to identify local urban policies that have addressed the issue of gentrification while implementing smart growth policies and whether they can be easily adapted to other cities.

### 9.2 Relationship and relevance of D1.3 to future RAMSES work

The research conducted by T1.3 and reported in this D1.3 are of relevance to various ongoing and future RAMSES work. For example our findings on national policies and city strategies (Chapter 2) may impact on the current and future RAMSES research in Tasks 5.4 (cost assessments) and 7.3 (policy tools). Also transition, stakeholder and toolbox developments from WP 8, 9 and 10 will benefit from these findings as city representatives, urban planners and EU policy makers can find qualitative and quantitate evidence on the influences of policies on city strategies and plans.

In Chapter 3 we evaluated different assessment methods, which will help to test alternative transition models in RAMSES WP8. The DEA can also be of use in the training and toolbox work of WP 10 as it helps to evaluate a set of adaptation options and identify contributing factors of energy efficiency across cities. The inventory we set out in Chapter 4 will be used in the estimation of adaptation cost curves to be included in D5.3, an approach presenting economic cost and benefits of adaptation options and their effectiveness.

The Performance Index (PI) from Chapter 4 provides a useful assessment and selection method which will help city planners to select the most suitable species in order to maximize the mitigation and adaptation potential of GUI vegetation. Thus it informs the transition alternatives testing (Task 8.3) and toolbox and training on climate change adaptation and sustainability (Task 10.1).
Investigating the impact strategies have on the uptake of EV in Chapter 6 has highlighted the importance of incentives such as infrastructure provisions and clear signalling by authorities, which informs the ongoing RAMSES research on transition (WP8) and stakeholder dialogues (WP9).

That urbanisation may drive climate change in developing countries and may mitigate climate change in developed ones as reported in Chapter 7 is of relevance to the wider research work of RAMSES and WP on transition. The findings of Chapter 8 confirmed that one can no longer view a city in isolation with clearly defined fictitious boundaries but rather as a dynamic cluster of entities constantly evolving in space and time, which has implications to the ongoing policy tools research in T7.3, transition alternative testing T8.3, and to the stakeholder dialogue as part of T9.2.

9.3 Final conclusions

To finally conclude, in this report we have benchmarked the vulnerability, adaptiveness and efficiencies of cities, and developed various inventories to assist researchers, urban planners and city representatives to assess individual city measures, infrastructure components and strategies. We plan to verify our results in future research based on updated city inventories as they become available. The research has demonstrated the complexity of an urban system and clearly demonstrated attribution costs and comprehensive assessments. We have developed and tested various aggregated or reduced forms of cities and infrastructure component using case studies that identify key elements, methods and strategies and, combine sector-based assessments with those more regionalised and localised. This has enabled us to determine key climatic impacts and to study certain types of infrastructural components, i.e. physical and social and assessed in terms of the usefulness and resilience in quantitative research terms. The report and our research promises not only to improve assessments but prompt the implementation of infrastructure components that support climate change adaptation strategies.
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- 103 -


*The United Nations Framework Convention on Climate Change.*


