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Concepts and Methodologies for Measuring the Sustainability of Cities

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Keywords greenhouse gas inventories, consumption-based, carbon footprint, environmental footprint, urban metabolism

Abstract:

Over the last decades, better data and methods have become available for understanding the complex functioning of cities and their impact on sustainability. This review synthesizes the recent developments in concepts and methods being used to measure the impacts of cities on environmental sustainability. It differentiates between a dominant trend in research literature that concentrates on the accounting and allocation of greenhouse gas (GHG) emissions and energy use to cities, and a re-emergence of studies focusing on the direct and indirect material and resource flows in cities. The methodological approaches reviewed may consider cities as either producers or consumers and all recognize that urban environmental impacts can be local, regional or global. As well as giving an overview of the methodological debates, the implications of the different approaches for policy are examined, as well as the challenges they face in their application on the field.

1 Introduction

Over the last decades, urbanization has grown at an unprecedented rate. The urban population is projected to reach 6.3 billion by 2050, from 3.4 billion in 2009. This is the equivalent of almost 70% of the world's population (1). These rates imply that virtually the entire world population could be urbanized by the end of the century (2). Most of the expected growth will be concentrated in Asia and Africa[1]. Although large cities will continue to grow at a fast pace, the bulk of new urban growth is predicted to take place in smaller urban centers (3). Current urbanization departs from past trends not only in terms of scale, but also in terms of the distribution of the urban population (4), as well as urban form and function (5). Forecasts for new urban land cover on a global scale vary depending on assumptions, but by 2030 new urban land could increase by an area roughly the size of Mongolia (6), or four times of the size of Germany.

There is no evidence to claim that urbanization *per se* is to blame for high levels of resource use and environmental degradation. However, there is a clear overall association between a country's economic growth and urbanization (7), and between the level of urbanization and industrialization (8). The increased consumption that arises from increased income is, therefore, where the link between urban living and environmental impacts can probably be drawn (9–11). Nations and cities now both recognize the potential for cities to reduce their impacts on the environment (12). For such potential to be realized, however, the first step is to establish reliable information on what the impacts are, where they originate and how they differ across urban areas. The purpose of city sustainability assessment methods may be, among others, to monitor current performance and to assess the impacts of measures; to prioritize the environmental aspects, locations or sectors in which to take action and support the design of policy solutions; to provide a basis for future scenarios; or to shed light on the underlying drivers and dynamics of city sustainability, i.e. to understand which processes, technologies or actors lead to which impacts.

This review addresses the concepts and methods used to measure the environmental impacts of cities. It focuses on two interconnected strands of research: on the one hand, a dominant trend in research on the accounting for and allocation of greenhouse gas (GHG) emissions and energy use to cities (also often called carbon footprinting) and, on the other, a re-emergence of studies focusing on urban metabolism or, in other words, the material and energy stocks and flows through cities. Both fields share common methodological aspects. In particular, both may consider cities as either producers or consumers and both recognize that urban environmental impacts can be local, regional or global. These concepts are central to this review and we elaborate on how they affect the perception of responsibility for concrete impacts, as well as their implications for policy. The two fields also show considerable divergence, especially regarding the degree of application of the existing knowledge on the ground. The reasons for this are examined and the main knowledge gaps are identified. Finally, recommendations for further integration of the two fields are given.

Research into the effect of urbanization on the environment comprises other aspects that are not covered here, such as land use change (see, for example (4)), loss of

[1] Note to editors: Data in the previous three statements could be updated with 2013 revision of UN's urbanization prospects, to be published in the first half of 2014. Update still not available as of March 2014, I will keep checking.

biodiversity (13, 14), resource scarcity, or the high vulnerability of cities to environmental stresses such as climate change (15, 16). The vast field of scenario modeling for cities is also not the focus of this review, although other authors (17, 18) do look into this, and the accounting of present and past impacts – which *is* the focus of this review- is a crucial basis for doing prospective analyses. Lastly, this review only touches upon social and economic sustainability aspects where relevant in the discussion of potential synergies with GHG accounting and urban metabolism methodologies (see "sidebar").

2 Overview of methodological approaches and conceptual considerations

The methodological approaches for quantifying GHG emissions and metabolic flows in cities are similar and the two fields are closely interlinked. Assessment of urban metabolism includes, in principle, discussion of GHG emissions and energy flows. While both fields have been reviewed in a joint manner elsewhere (11, 17, 19, 20), in this review we chose to highlight the narrower domain of energy and GHG-related accounting as a separate field of research (or a sub-field within urban metabolism research) that has a much wider uptake in practice. We believe that by separating the two bodies of knowledge we can help to build a picture of the strengths and weaknesses of each field and shed greater light on how best to bridge the gap between them, in order to address the assessment of urban environmental sustainability in an integrated way, a need that has been highlighted by others (e.g. (19)).

Views on how the accounting methods to measuring the environmental impacts of urban systems should be classified differ; however, the conceptual approaches are broadly three (see Table 1 and/or Figure 1):

- The **territorial** or production-based approach quantifies the GHG emissions produced, or the energy and materials used, within a geographic or jurisdictional boundary. For example, it quantifies the amount of fuel used by private households as well as by industrial facilities, power plants and transport within a city and their associated GHG emissions. Local (and national) energy balances and emission inventories use this approach.
- The **supply-chain** approach is typically used to complement territorial inventories as it also measures the – often significant – impacts of urban resource and energy use along cross-boundary energy and infrastructure supply chains. For example, it accounts for the upstream GHG emissions of electricity or of other resources such as water, steel and cement imported from other locations to serve local consumption, as well as the downstream GHG emissions from the use of a waste treatment plant that is beyond the city boundary.
- The purely **consumption-based** approach, which focuses on capturing all the global impacts of the final consumption in a city. This is achieved with environmentally extended input-output tables (EEIOTs), which take the full set of interactions within the economy into consideration. This differs from the other approaches because it only takes into account the impacts of the *demand* for services of a city's private and public final consumers. In other words, productive activities within the city which satisfy demand in other locations are not taken into account.

The refinement of territorial approaches with supply chain-based approaches and also the emergence of a consumption-based perspective over the last decade are not unique to cities, as these trends have also developed in the context of national GHG emission and resource accounting (21–25).

A set of conceptual considerations arise from this classification of approaches:

- ***Responsibility***: these different views of how city impacts may be measured – either production or consumption-based – are linked to the capacity of different sets of actors (local governments, state governments, consumers) to act on the impacts and sources and, therefore, on the allocation of responsibility. There is broad consensus that different methodological approaches can address distinct policy levers in complementary ways.
- ***Boundaries***: a fundamental problem in all fields is the definition of the city's boundary. Boundaries may be set according to administrative limits - which often determine the areas for which statistical data is available - land use, geographic, economic or other criteria (e.g. commuter sheds). A pervading limitation for comparison is the fact that there is no universal consensus on what constitutes an urban area or a city (e.g. (1, 3)).
- ***Data:*** despite the acknowledgement of the importance of assessment, access to reliable, complete and comparable data remains a challenge – to varying degrees – for all areas of the matrix shown in Table 1. The accuracy problems are resolved in different ways (e.g. downscaling, interpolations). At present the standardization of data collection protocols is being attempted in both energy and GHG accounting and urban metabolism, as discussed below (26, 27).

In the following sections, these conceptual aspects are discussed in the specific context of the existing literature for energy use and GHG emissions accounting and for the urban metabolism fields.

3 Energy use and GHG emissions

Since the 1992 UN Conference on Environment and Development in Rio and its resulting "Local Agenda 21", bottom-up energy use and GHG emission accounting by cities has established itself. Organizations such as the U.S. Mayors Climate Protection Agreement, the Climate Alliance, the Covenant of Mayors and ICLEI-Local Governments for Sustainability have supported the implementation of inventories in thousands of communities worldwide (28–30). In 2011 the Chinese central government issued "Guidelines on Provincial Greenhouse Gas Emission Inventory (Trial)" which also serves as a basis for developing city inventories (31), while it is reported for Germany that the majority of larger cities have been compiling GHG inventories for over a decade and many update them on a regular basis (28). Moreover, various support tools for carrying out local inventories have been developed (32) (see Table 2).

The many inventories that have been carried out are often referred to as "carbon footprints" (CF) . As there is no clear and uniform definition of this term, here we designate CFs as either territorial, supply chain or consumption-based CFs to differentiate between the different possible methodological approaches to carbon footprinting. Historically, most inventories initially took a purely territorial (or production-based) approach but were gradually extended to include sources beyond their boundaries (33–35). This extension typically started with cross-boundary electricity generation for consumption within the city, as well as upstream emissions

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from fuel use and downstream emissions from waste. Later, some broadened into partial or full consumption-based studies. The different approaches are now seen as complementary by many (36–39) and several comparisons have been carried out on the quantitative and qualitative scale – these are reviewed below. It is important to highlight that in addition to these main approaches there is an adjacent field of research that attempts to measure the GHG emissions of metropolitan regions directly from terrestrial and satellite measurements (40). The following sections discuss the practical implementation of GHG inventorying, the current challenges faced and the consequences for policy decision-makers.

3.1 Main approaches

Early efforts towards a uniform methodology for inventorying GHG emissions were laid out by Kennedy and colleagues (41), who combined the strictly territorial IPCC GHG emission inventory guidelines (42) with the scope approach of the World Resources Institute/World Business Council for Sustainable Development (43) for cross-boundary impacts, in an attempt to develop a conceptual framework for the context of cities.

Recently, impetus in this direction came through the establishment of the Global Protocol for Community-Scale Greenhouse Gas Emissions (GPC) (26), jointly created in 2012 by the WRI, C40 and ICLEI, in partnership with the World Bank, UNEP and UN-HABITAT. The GPC protocol intends to address the criticisms on the differing approaches between cities (44) and to replace earlier protocols. In 2013, 35 cities from around the world started pilot-testing the GPC. The precursors of the GPC are now seeking international consensus on methodologies for the accounting of cross-boundary emissions, and are aiming for the protocol to be adopted as the standard for cities by 2015. This is, therefore, a crucial moment in the debate.

Table 4 gives an overview of the correspondence between the GPC's "scope framework" (Table 3) and that of the three methodological approaches shown in Table 1.

Territorial approaches are mainly found in local and other initiatives such as the GPC and its predecessors (45, 46). Strict territorial approaches (scope 1 in the nomenclature of the GPC) do not reflect the emissions outside city boundaries and they usually rely on local statistics, and therefore typically refer to an administrative territory for which data is available to local authorities. Territorial scope 1 GHG inventories have the advantage of being compatible with IPCC definitions at national level (see Table 2). Moreover, they avoid the risk of double-counting (47), whereas supply chain and consumption-based approaches (scope 2 and 3 in the nomenclature of the GPC) generally do not. For instance, when emissions from electricity consumption are calculated using the emission factor of the grid, local production should be ignored in the balance to avoid double-counting. In the case of studies which combine a consumption-based approach with scope 1 emission inventories, the problem becomes almost unsolvable as parts but not all of the goods and services consumed are produced within the city's territory (48).

The territorial emissions of cities vary according to the definition of the boundary used; administrative city boundaries can be as narrow as the central business district (e.g. in North American or Australian cities) or include a city region with a large share of the hinterland as is typical in the administrative boundary of a "city" in China (49–51). Consequently, the geographical scope used has significant implications for comparisons and benchmarking across cities. Cai and Zhang (52) exemplify this effect with a case study in the city of Tianjin: while the densely inhabited six central districts of Tianjin account for more than a third of the population, per capita scope 1 and 2 emissions are 60% below the average for the whole city region due to the predominance of the service sector over manufacturing. This example also shows that the geographical boundary influences the relative shares of scope 1 and scope 2 emissions. Emissions from "imported" electricity are as high as one third (35%) of total emissions for the central city but are only 12% for the city region. Wider geographical boundaries (i.e. the administrative city region as opposed to the central district only) thus tend to result in larger shares of scope 1 emissions from power generation and industry (and of waste treatment, which was not analyzed in this particular study) as well as in a lower prominence of imports in the inventory.

A core problem of compiling territorial GHG inventories and supply chain emissions is the availability of local data for energy use, transport and other relevant activities. Accuracy, data collection frequency, whether or not the data is up to date and, sometimes, confidentiality and processing costs are all pertinent issues (45). Table 2 shows a selection of five software tools that have been developed to support local authorities in compiling their GHG inventories and baselines. These tools are already used by many cities, mainly in Western European countries. Although most of the tools basically follow a territorial approach, there are still differences in the details, e.g. coverage of non-energy GHG emissions and treatment of emissions from mobile sources, reflecting local inventorying practice in the countries where the tools are most used. As a result no internationally accepted tool is available as yet (45).

To overcome omissions in data at the local level, local authorities often use estimates derived from national statistics to complement their actual data; for example, many cities lack data about the consumption of oil, coal and other fuels not delivered via dedicated infrastructures (i.e. not connected to the natural gas or district heating grid). To estimate these data, sales statistics or typical consumption figures for households and industries are used (28). Most of the tools presented in Table 2 provide national averages to help estimate energy use where data is not available from local statistics or energy suppliers. This is the case of, for instance, the widely-used ECO_2Region tool (45, 53).

Mobile sources pose a particular problem for data availability in territorial approaches. For example, passenger transport energy use and emissions can be estimated using fuel sales data and/or vehicle kilometers traveled – if it is available and if there is relatively little transport across city borders (41, 48). Smaller cities – in which most transport occurs across city boundaries – often use consumption-oriented approaches to estimate data, for example multiplying total travel by residents by emission factors per vehicle kilometer (41). Sometimes commuters are also explicitly accounted for (36). In large cities, emissions from (mainly international) air- and water-borne transport can be very significant, as illustrated in the examples of Hong Kong (47), London, New York, Geneva and Cape Town (33), with estimates derived from territorial fuel sales. Kennedy and colleagues (54) provide further details on what is and what is not included in the inventories of 44 urban areas, including in developing countries.

Supply-chain approaches (or "extended territorial" approaches (39)) using life-cycle data are frequently used to refine territorial inventories, in order to take account of the often significant emissions resulting from energy and other infrastructure goods. The trans-boundary Community-Wide Infrastructure Footprint by Chavez and Ramaswami (36) is a good example of a comprehensive supply-chain approach. For data, they rely either on the specific supply structure of a city (e.g. purchases of energy from certain power plants via long-term contracts as reported for Hong Kong (47)) or on emission factors for the relevant regional or national electricity network of which the city is part (55). When comparing studies for ten global cities, Kennedy and colleagues (33) found that "full life-cycle" emissions (territorial plus supply-chain, and partly including non-energy infrastructure imports) ranged between 4.6 and 24.3 tCO_2e/cap, with the three North American cities (Denver, Los Angeles and Toronto) being the top emitters. Supply-chain emissions in the ten cities were typically about as high as territorial emissions, with the exception of Geneva and Toronto, which both have very low supply-chain emissions due to very large shares of hydro electricity supply. Including upstream emissions from heating, tansportation and industrial fuels increased the GHG emissions attributable to cities by between 7% and 24%. In the case of the top emitter, Denver, that increase translated into a further 2.8 t CO_2e/cap. Furthermore, if non-energy infrastructure emissions are included (i.e. embodied emissions for food and cement), Denver's emissions increase by an additional 2.9 t CO_2e/cap (35).

Downstream emissions from waste treated outside city boundaries (scope 3) are often also covered in the supply-chain approach. For these, studies either calculate the emissions of the city's share of the use of the waste treatment plants and landfills that are beyond its boundary, or use typical emissions values for waste per capita (19). The latter however requires important assumptions (56), with the methane recovery of landfills being the most important factor: emissions from waste can be significantly above 1 tCO_2e/cap when methane is not recovered, but significantly lower with methane recovery (33).

Many of the available supporting tools (cf. Table 2) also provide life-cycle data, e.g. emission factors of national electricity systems or global upstream emissions from the supply of different fuels (45).

Consumption-based approaches are frequent in scientific literature (17, 24, 57). They typically assign emissions to the sectors of final consumption (households and public administration) in a city's territory, by means of EEIOTs (24, 48). The GPC terms these approaches as scope 3 (see Tables 3 and 4). ‚One disadvantage of input-output tables is that most are usually available only at national level. They therefore do not allow a distinction between territorial direct emissions and national and international indirect emissions. For this reason, consumption-based approaches are sometimes combined with others. When they are combined with territorial approaches, problems of double counting can occur (see above). They can also be supplemented with more detailed analyses, which are similar to supply-chain approaches, e.g. considering emissions from electricity demand (e.g.(39)) or from transport and residential heating. Some studies check for accuracy against energy statistics or against GHG inventories for a small subset of municipalities (e.g. Finnish municipalities (57)).

Consumption-based studies combine the use of EEIOTs with data on income spending from micro-census data and consumption surveys to calculate global impacts from local final consumption. Such data are often provided per municipality (57, 58) or even on smaller spatial scales (38). When such data are available for a whole country, urban-level studies can be carried out for all the municipalities of a country (39, 57–59) using the same methodology and data. However, the focus on final consumption within a city is a significant limitation to the approach. Based on consumption-based data from 20 US cities (all of which had submitted a GHG emission inventory to ICLEI), Ramaswami and Chavez (48) demonstrate that the carbon footprint mainly reflects income and wealth or, as they put it, the "*willingness of different cities' residents to consume*" but is less indicative of the specific infrastructural situation in a city.

Another disadvantage of consumption-based studies is the heavy reliance of input output tables on national or even global data that are defined in monetary (i.e. not physical) terms, that are often highly aggregated (60) for economic sectors and reflect neither the characteristics and emission intensities of local production for domestic use nor the specific nature of the relevant supply chains (36, 48, 60). Studies often acknowledge that IO tables use historic data and often may not reflect the current situation of a city (61). Furthermore, some studies simply assume that imports reflect national or US production functions (57, 60). Recent developments in EEIO analysis partially address these difficulties as IO tables are produced more frequently and with higher levels of disaggregation including physical values (61).

International multi-region input output analyses (MRIO) allow for more accurate quantification of the environmental impacts embodied in international trade (21, 62, 63). However sub-national MRIO data that reflects local production functions is rarely available (36). As an exception, Lenzen and Peters (64) do assess the impacts of households, including GHG emissions, in two Australian cities (Sydney and Melbourne) based on MRIO data at state level. Ramaswami and Chavez (36, 48), also assess the GHG emissions associated with household consumption using downscaled IO data for several US cities. However, accuracy remains particularly challenging for subnational MRIO. Input-output data downscaled from the global or national level is associated with high uncertainties, as it does not, for example, reflect actual energy flows associated with energy use at local level (36, 65). In addition, the integration of physical and economic data at subnational level is difficult, because these statistics exist in varying degrees of detail for different industry sectors (64). Furthermore, Wiedman and Barrett (62) claim that the resolution of MRIO remains low at product level. To overcome this, they propose the systematic combination of IO analyses with life-cycle analyses.

Having reviewed the main methodological approaches to GHG emission accounting for cities, the next section will discuss ways in which the results of GHG accounting exercises can be aggregated to give a picture of city typologies, which can be both of methodological and policy relevance.

3.2 Typology of cities according to emission profiles

Cities vary widely in population, size, density, structure, economy and relationships with their hinterland and other regions. Additionally, different world regions show substantially different patterns of urbanization and use different parameters to define a city. Despite the huge challenges imposed by this diversity, some STUDIES have tried to develop categories for cities based on their specific energy and emission profiles.

Chavez and Ramaswami (36) attempt to establish **three types of cities,** mainly reflecting their economic structure and based on data for US cities that span the three methodological approaches mentioned above. *Net producers* have higher territorial emissions compared to their consumption-based emissions. Since they are net exporters of emissions, a significant share of emissions from their territory is attributable to consumption in other cities. The opposite is the case for *net consumers*, while for *trade-balanced* cities the balance of emissions in the territory and emissions attributable to final consumption in the city is more or less equivalent. Schulz (66) however shows with the example of Singapore that cities can shift to a different category over time, or even frequently. In Singapore, indirect CO_2 emissions embedded in imports as well as in exports reached per capita levels as high as 5 to 15 tCO_2/year in the 1990s and the city's net balance has since been fluctuating between negative and positive – making it a net importer in some years and a net exporter in others.

Though not explicitly referring to these city typologies, some empirical studies give some hints on the prevalence of the different types: For example, a comparison of (consumption-based) carbon footprint estimates with 'extended' territorial CO_2 emission estimates (scope 1 and 2 emissions) for all UK municipalities found that most of them are net consumers– due to the fact that the UK as a whole is a net consumer (39). In Finland and the UK, net producers have been found to be mainly municipalities hosting heavy industry or large power plants, many of which are rural (39, 67).

Melbourne represents a rather unique case for a developed country city, where territorial energy use was estimated to be up to 10% higher than consumption-based energy requirements, as derived from household expenditure (38). This result can be explained by the significance of the local manufacturing, petroleum and tertiary sectors. So-called 'gateway cities' are another type of net exporting city. Gateway status is often determined by a city's location as well as by having a central function as a site of a major port or airport. As examples show (33, 47), emissions from aviation or shipping for such gateway cities are attributed by measuring fuel sales.

In addition to economic structure, location, urban form and technology are determining factors for urban energy use and emissions (33, 41). Particularly urban form, which to a large extent determines local transport emissions and technology (e.g. waste treatment emission abatement technologies but also electricity supply, quality of building and efficiency of cars etc.) are important (33) but not always within the scope of local policy. The wide scientific discourse around these topics cannot be discussed in depth here.

3.3 Implications for policy

Urban energy and GHG emission accounting is aimed at managing and improving on urban as well as global impacts of cities. With this common purpose, territorial, supply chain and consumption-based emission profiles give different insights which provide complementary information for decision making (48).

Consumption-based approaches focus on the final consumption of goods and services by resident households (and public administration) and the corresponding global impacts. They are, therefore, of greater relevance to managing consumption behavior and consumer responsibility and concern increasingly global debates on sustainable production and consumption (17, 38, 48). As local consumption-based estimates are typically derived from national data that are available for many or all municipalities based on comparable methodology (as in e.g.(39)) they are seen as appropriate for benchmarking and for the purposes of raising society's awareness (36, 59).

However, consumption-based CO_2 emissions of municipalities tend to be much more homogeneous than territorial emissions. For example, in their study of the UK, Minx and colleagues (39) show that, in contrast to territorial (scope 1+2) emissions which range from 4.3tCO_2/cap to 60tCO_2/cap, the consumption-based estimates all fall into a narrow range of 10-15tCO_2 per capita (reference year 2004), with no overall regional pattern emerging. This puts into question the usefulness of consumption-based benchmarks. Another criticism is that local decision-making based on consumption-based estimates has its limitations, most pointedly, in the area of housing and transport (39), which are on the other hand often the most important sources of private households' GHG emissions.

The local economic structure, the management of the metropolitan economy and infrastructure (in-boundary and trans-boundary), transport and energy systems – as well as the potential transitions of these – are better supported by **territorial and supply-chain approaches** (48, 68). These approaches provide greater insight into local production and infrastructure characteristics and are therefore recommended for informing local and regional infrastructure planning for GHG mitigation. In particular, territorial direct emissions accounts inform best about local impacts such as air quality, micro-climatic or health effects. Supply-chain data for key infrastructures is suited to informing about relatively direct effects from the sourcing of energy, water and other resources for the city, the concrete impacts of the disposal of waste and wastewater, as well as the vulnerabiltiy of the supply chain and related planning (36). Furthermore, data quality is thought to be better in (extended) territorial approaches, and local inventory data are also more suitable for combining with national inventories, as harmonized methodologies that avoid double counting are in place (47).

Given these distinct characteristics of the three approaches, the current practice of protocols used at local level (69) seems to be well reasoned, as they start by harmonizing the territorial and supply chain approaches, and in some cases they proceed to complementing this basis with consumption-based EEIOT data to provide information for consumption and behavior-oriented policy approaches.

Finally, monitoring of cities over time is another area in which GHG accounting can support policy. Kennedy and others (44)[2] analyzed emission trends for six cities which repeated their GHG inventories over a 5-year period: Berlin, Boston, Greater Toronto, London, New York and Seattle. They discovered that, in line with national patterns, the GHG emissions of these cities are declining (by an average 0.27 tCO_2e/cap/annum), despite population growth. This is as a result of changes in the patterns of stationary combustion, as well as through a general reduction in carbon intensity via fuel switching. A study of 30 German cities with over 100,000 inhabitants that have compiled CO_2 emissions over an average of 12 years found that they reduced their emissions by 1.31% per year on average (with population remaining generally stable). This was slightly below the national average of 1.35% per year (28), but there was huge variation between cities, with the highest reductions occuring in East German cities due to the restructuring of the economy.

3.4 Future challenges

A solution has yet to be found, however, for a number of issues. These are:

[2] Note this study is not a strictly a comparison of methodological approaches but of the challenges in the application of inventory guidelines as well as value of time series, which can also be considered of methodological relevance.

- Data quality and availability issues, linked to limited local capacities (44, 55, 70) in both developing country and also developed country cities.
- The relationship between urban land cover and carbon losses from land use change is rarely studied. Seto et al. (4) use spatial probabilistic models to show that the conversion of land to urban land is likely to become substantially more important to emissions from deforestation and land-use change than previously evaluated.
- In spite of much work on urban GHG inventories which have been carried out for decades (e.g. in Germany (28)), there is still a lack of differentiated and reliable comparisons between cities and, equally, of well- founded time series. Such data would inform about occurring trends and help to identify drivers and key sources of emissions.
- Research tends still to focus on large metropolises, which is understandable as these provide the most comprehensive datasets. However, this is concerning considering that most urban growth is expected in mid-size cities.

Additional open issues beyond the scope of this paper include analytical and prospective modeling approaches needed to further understand drivers of change and levers for policy. Moreover research is needed on how models can be best based on urban GHG emission accounting, as well as their role in urban and multilevel governance (17, 71, 72).

4 Material, energy and substance flows

Wolman (73) was one of the first to apply the concept of urban metabolism (UM) by calculating the inputs of water, food and fuels and outputs of wastewater, solid waste and air pollutants for a hypothetical American city. Since then, the UM framework, which is rooted in concepts and models from ecology, has been applied as a means of accounting for the energy, water, nutrients, materials and waste that flow in and out of urban areas, as well as those that are stored or embodied in the built environment (74) (Figure 2).

There has been a re-emergence of UM studies since the 1990s, which have recently been reviewed by several authors (17, 75–78). Moreover, recent years have seen increased debate on the improvement of city sustainability assessment through the integration of UM models with life cycle analysis, urban quality measures and other related fields of research. The main approaches, specific methods and current debates are reviewed here. The field of energy and GHG-related accounting can in fact be considered a sub-field within UM research, although it is not always acknowledged as such. This narrower field has been discussed separately in the section above due to the fact that it has a much wider uptake in practice.

4.1 Main approaches

Three main methods can be discerned in UM literature:

- **Material Flow Analysis (MFA),** which accounts for flows and stocks of either single materials or sets of materials, in terms of mass[3]. Various indicators are used; those accounting for flows are usually expressed in tons per capita per

[3] An overview of the material flow based types of analysis across various scales is given by Bringezu and colleagues (79)

year and those for stocks are expressed in tons per capita. For example, the indicator for yearly domestic material consumption (DMC) in Paris ranges from 2.2t/cap to 5.0t/cap, depending on whether the export of waste to the surrounding region is considered (80). MFA has been the most favored method in the UM literature, which varies considerably in terms of the scope of the material flows considered. Some studies make a comprehensive material balance, such as Vienna (81, 82), Hong Kong (83, 84), Hamburg and Leipzig (82), Singapore (85, 86), Paris (80), Lisbon (87) and the greater Toronto area (88). Kennedy and Hoornweg (27) identify twenty such comprehensive studies of the metabolism of a specific city. The remaining literature predominantly takes one particular material or flow as a focus for analysis. This includes, for example, urban water balances (89, 90). Moreover a significant body of studies that quantify the energy use and/or GHG emissions of cities, as covered in section 3, use an MFA framework. Changes in urban metabolic processes over time have also been analyzed: Kennedy and colleagues (74) collected UM data from eight major cities and complemented it with studies of particular flows for other cities, showing an upward trend in per capita resource inputs, energy intensity and waste outputs in cities such as Hong Kong, but also varied patterns in other cities, with efficiency increasing in some. They also draw attention to the use of UM for understanding changes in material stocks, although the emphasis in MFA is often put on the flows (74). A complementary strand of MFA research focuses on the dynamics of urban building stocks (e.g. (91, 92)).

- **Energy flow analysis** quantifies the energy flows of all types of energy carriers jointly, based on their enthalpy content. Emergy analysis is a particular type of energy flow analysis which quantifies both energy and mass flows jointly by converting them into a single comparable unit termed solar energy joule (seJ). Thus, emergy analysis allows the comparison of flows of different *quality*, recognizing for example the different energy intensities of the various materials consumed in cities. Although not mainstream in the UM literature, this approach is now increasingly common (e.g. (93–95)). Methodological comparisons between emergy analysis and MFA have been carried out by Huang and colleagues (96, 97) who illustrate some limitations of MFA and advocate for a stronger link between emergy and material methods.
- **Substance Flow Analysis (SFA**) is concerned with individual chemical elements or sets of elements and is, therefore, used to address more specific issues such as air or water pollution or nutrient accumulation in cities. For example: quantifying the flow of phosphorous discharged into water bodies (98) or the nitrogen fluxes in a city's food metabolism (99)..

As in the case of GHG accounting (cf. section 3), MFA and other systems'-based methods for urban sustainability assessment have been used on other system scales, predominantly at national and household level (100–102). Economy-wide MFA is relatively standardized and methodological guidelines exist (e.g. by Eurostat), whereas at the urban level this is not yet the case. Similarly, an evolution from production-based to consumption-based approaches (cf. Table 1) is observed in the UM field. Traditionally, UM studies focused largely on the processes that sustain cities and not on the impacts that cities cause on distant or global processes (103, 104). There is a value in this, as resources such as water, or outgoing waste flows, often retain strong links to the areas surrounding cities and self-reliance can be of high policy priority (90). An understanding of the relationship between a city and its hinterland – or historical "supplying" area – can be achieved through a territorial approach. However, the emergence of global trade in industrial materials and products, and of concentrated centres of manufacture, has led to the recognition of the importance of the relationship of a city to its "global hinterland" and of methods that can show the delocalization of urban resource consumption and of emissions and waste (103, 105).

Several UM studies have taken a supply-chain or a consumption perspective in order to account for the impacts of trans-boundary urban metabolic flows. This recent literature uses a combination of approaches, such as EEIOTs (discussed in section 3), household consumption surveys, or Life Cycle Analysis (LCA) data, in addition to the bottom-up mass or energy balances provided by UM models.

Environmental footprints (EF) provide one such way of quantifying the impacts of urban metabolic flows across urban boundaries (77, 78, 104, 106). Examples include urban water footprints, or the total volume of water used in the production and consumption of goods and services in cities, as well as direct consumption. Water footprints at urban or municipal level are not yet common, although some regional studies do exist (107, 108). These sometimes contain high levels of spatial detail, pinpointing in certain cases to whether the consumption-based footprint affects water-scarce locations (108). Using notions related to water footprints, Lenzen and Peters (64) analyzed the water requirements that inner-city households in two Australian cities posed on their national hinterland, showing that the indirect use of water was four times higher than the direct use. This is an example of the use of a spatially explicit model, possible only with the rich multi-region input−output (MRIO) databases available in certain countries (see section 3.1). Spatially explicit studies of this type at urban level remain relatively rare (76).

Ecological footprints have seen relatively more use on the urban and regional scale, both in academic literature and from the perspective of local government institutions (107, 109–114). Ecological footprints calculate the amount of land (usually in terms of "global hectares") required to meet a territory's needs in terms of resource supply and waste disposal (115). The studies reviewed by Kennedy and colleagues (74) show that "*the areas of ecosystems required for sustaining cities are one or two orders of magnitude greater than the areas of the cities themselves*". The types of land can also be identified: for example, a recent study of metropolitan Vancouver's ecological footprint shows that carbon sink ecosystems (forested land required to sequester GHG emissions) and cropland for food growing comprise 90% of the city's footprint (113). The ecological footprint indicator is considered to be well suited for education and for purposes of communication to policy makers; however, its adequacy for guiding policy is contested (62, 106, 116). Wilson and colleagues (114) calculate neighbourhood-level ecological footprint values for 241 neighbourhoods in Oakville, Ontario, resulting in an average ecological footprint for the town of 9 hectares per capita. The detailed energy and material flow information at neighbourhood level provides a basis for suggesting a potential "footprint floor" for Oakville of around 5 ha/cap and for guiding the changes needed to reach this target.

One of the criticisms of UM methods is that their functional units (e.g. energy, mass or land units) do not capture the varying pressures that different flows can have on the environment. For this reason the use of hybrid methodologies that combine LCA with UM is being proposed (78, 104, 117, 118). The use of LCA data also helps to elucidate cross-boundary effects of urban metabolic flows. In practice, and in contrast to the relatively wider use of LCA data in the context of supply-chain approaches to city GHG accounting (65), the coupling of UM with LCA for cities is scarce (78, 119). Goldstein and colleagues (78) test a pragmatic application of LCA categories on existing UM datasets for five global cities. The UM-LCA model shows that the mass flows corresponding to upstream and end-of-life stages of cities' metabolic activities was considerably higher than the “use stage” that is typically considered within UM. An exception was Beijing, where direct mass flows accounted for 75% of the total, a result attributed to its large consumption of concrete. The results not only illustrate how the coupling of UM with LCA can account for cross-boundary impacts, it also complements the insights provided by UM data on specific environmental pressures exerted by cities. The integration of LCA and UM also has its drawbacks: LCA methods are not free from uncertainty and subjectivity (an example is the assumptions needed about the timeframe during which potential impacts take place), so it could be argued that UM remains a useful tool on its own.

The issue of where to draw the boundary of a city is equally as important in UM as it is in the energy use/GHG emissions accounting field. Currently, UM studies define city boundaries in different ways, such as commuter sheds, densely populated/built-up areas or jurisdictional boundaries. A number of studies compare the effect of the different spatial scales. For example, in Paris (80) the food consumption per capita is higher in the dense city center because of the large number of employees and tourists, whereas large amounts of construction materials and fuels are consumed in the outer suburbs due to new housing and infrastructure. This kind of result shows the link between UM studies and land use planning.

4.2 Implications for policy

Despite the strong theoretical backdrop and the existence of applied research, the use of UM methods for decision support in the field is still scarce. Some cities have used them in the past for defining targets for dematerialization or improvements in material efficiency. MFA data can also be used for estimating urban mining stocks (120, 121). The lessons learnt from UM have been applied in urban design; for example, the Hammarby Sjösted urban district in Stockholm, Sweden, which is often cited as an exemplar in sustainable urban development, was based on the principles of circular metabolism (122).

Kennedy and Hoornweg (27) identify a number of initiatives, such as a project funded by California Energy Commission, which is piloting the collection of spatially-explicit UM data in tandem with socio-demographic data for the county of Los Angeles (123). Moreover, they highlight two recent projects funded under the EU’s 7th Framework Programme (124, 125). Within the scope of the SUME project, a spatially explicit metabolism model was applied to four case studies (Vienna, Stockholm, Oporto, Newcastle) to analyze alternative long-term urban development paths (126, 127). As part of the BRIDGE project, five European cities were able to evaluate different planning alternatives using a pilot decision-support system which showed how alternatives could modify the physical flows of specific UM components. The users could provide weights to denote the relative importance of each sustainability objective (128). While both these field-focused projects complain of the challenging data collection aspects, they demonstrate that there is significant momentum for the study of UM that is relevant to planning and policy.

A key policy application that UM can provide is that of benchmarking. However, despite an acceleration in the research during the 2000s, there are still few cross-sectional studies and a lack of time series studies of UM (75, 106). One comparative analysis was carried out by Saldivar-Sali (129) who developed a typology of urban metabolic profiles for 155 cities based on country-level MFA data. The study was not intended as a precise measurement of cities’ material flows, but rather aimed to provide a first-order approximation of their magnitude and to enable the comparison of city types. It classifies the cities by level of TMC (total material consumption) and by the pattern of resource consumption (i.e. high/medium/low level consumption of eight resources, as well as CO_2 emissions), resulting in fifteen clusters which provide insights into the levers that cities might exploit to address their most important resource consumption challenges.

4.3 Future challenges

Extending the UM model further

Urban metabolism has attracted criticism for being static in space and time and for disregarding the inner dynamics of city consumption, as urban metabolic models tended to treat the city territory as a “black box”, which precluded the assessment of sub-systems within the city (78). There is, however, an increasing variety in the scope of research: often there is not just a quantification of the metabolic flows but also models that aim to elucidate the relationship between urban environmental impacts and technological, land use or infrastructural drivers of metabolic flows. An example in the urban planning field is the link of the design of different neighborhood types in Toronto, Canada, to metabolic flows (130). The use of UM for dynamic simulation exercises is also growing (17), but is beyond the scope of this review.

Moreover there are calls to further extend the UM models to socio-economic and political drivers, in particular stakeholders or agents that influence flows (77, 80, 131). The inclusion of liveability indicators into UM studies (e.g. social amenity, employment, health) has already been proposed for policy assessment in Australia (132), but no follow up to this has been found. A further avenue for research that is worth noting is the coupling of UM accounting with the analysis of urban ecosystem services (e.g. studies on ecosystem services provided by parks and open spaces, or of urban ecosystem processes such as watersheds) (118). An example of the practical application is in Huang and colleagues (133), who analyzed the role of peri-urban ecosystems in the greater Taipei area in contributing services to urban residents, showing the shift in the types of ecosystems that have provided the most critical contributions over the last decades, away from agricultural production areas and towards upstream watersheds.

Data availability and methodological comparability

In contrast to the GHG accounting field described above, studies for UM and environmental footprints of cities remain relatively scattered and data has been established for fewer cities worldwide. The main reasons for the challenges found in the UM field are the lack of a uniform methodology, the difficulty in defining the boundaries of urban areas, the lack of direct data at city level (which leads to top-down scaling and estimations), the complexity of full material and energy flow analysis, and the labor-intensity of compiling material balances at urban level (17, 75, 106). The intricacy of UM studies is also said to affect its communicability to policy makers (78).

Urban resource flows have been explored by UNEP, who consider the UM framework to be an important step for identifying opportunities for resource decoupling in cities (134). However, while metabolism and environmental footprint studies of cities continue to be carried out, there is – as yet – no comprehensive global effort to harmonize accounting protocols (cf. the GPC initiative, section 3), nor readily available accounting tools for cities (cf. Table 2). Efforts to streamline the development of required indicators are still to become established. As a result, urban areas cannot currently be monitored over time or compared to other urban areas, so the knowledge we have is relatively poor compared to that on the impact of urban structures on energy consumption and GHG emissions.

Several authors have recently put forward proposals for general UM indicators that all cities should be able to collect (see Figure 3). Kennedy and Hoornweg (27) propose a selection of basic UM indicators focusing on key metabolic flows for which the recently established Global Cities Indicator Facility could become a data repository (27, 135). They integrate this selection with the indicators which are required for GHG accounting for cities. A framework for implementation of the UM method, based on publicly available data, was also proposed in the European context by Minx and colleagues (106). They tested the approach on a selection of European urban areas of varying size and density and gave recommendations for the development of a database based on existing sources, in particular from the urban audit initiative set up under the lead of Eurostat and complemented by other databases. This framework focused on the areas of water, energy and climate change, land and waste and largely omitted indirect metabolic flows required in the production of goods and services consumed in cities. Minx and colleagues highlight that despite this pragmatic approach, it is still challenging to find public data sources for a large range of cities, even for energy and GHG emission data.

5 Conclusion: lessons learnt from city sustainability assessment methods and methodological challenges ahead

The assessment of the sustainability impacts that cities have at a local, regional and global level is emerging as a critical research area. This paper has reviewed existing methods and tools for measuring certain dimensions of urban environmental sustainability, focusing on those used in energy use/GHG accounting and urban metabolism (UM) studies covering material, energy or substance flows. Both areas originate from a systems approach – the UM field takes the city ecosystem as the fundamental unit of analysis and much of city GHG accounting literature applies the same notion. In this review we chose to highlight the narrower domain of energy and GHG-related accounting as a separate field of research (or a sub-field within UM research) which is applied widely in practice, whereas the analysis of material flows for cities is yet to become mainstream.

The early **city energy and GHG inventory analysis** took a territorial perspective. Much unpublished work can be found in this area, starting from the early 1990s. The scope of research then broadened and energy and carbon footprints, which go beyond the city boundaries, became increasingly common. Such trans-boundary effects are often reflected as an addition to territorial inventories. They use supply chain approaches covering different parts of the energy infrastructures and also include downstream flows such as waste management. A stronger focus on capturing the complete global supply chain of goods and services is made by consumption-based approaches, which use EEIOTs. These, however, only determine global energy and GHG emissions attributable to final consumption in a city's territory. The footprints created by local production and economy are not separated as they are seen as part of global supply chains.

Urban metabolism has inspired new ways of thinking about the environmental resources needed to maintain cities across different spatial scales and on how the use of resources can be made more sustainable. Cities can potentially use UM indicators in practical ways, such as building their material management strategies on the basis of their input-output ratio and degree of circularity of the material fluxes, deciding on which stocks of materials are suitable for replacing primary materials, or quantifying the magnitude of waste flows. Despite its longstanding history, UM is still fragmentary as a field of research. Specifically, consumption-based assessment of UM, outside the energy and GHG emissions field, is far from being a standardized approach. However, a few studies have attempted to estimate the full supply chain impacts of the flows of certain materials consumed in cities through the use EEIOTs and household consumption surveys (e.g. (64, 78). Recent years have seen a re-emergence of UM in general and a consensus on the need to create a dialogue between this and other disciplines that address the governance and social processes that form cities (131).

In this review, data from studies has been used to illustrate the importance of different methodological considerations, such as the territorial vs. supply-chain and consumption-based approaches, and the role of the urban system's boundary. As a consequence a set of considerations for the future arise:

- As with any other measurement method, the ones reviewed cannot provide a magic formula to make cities sustainable. They can however aid in the understanding of policy options by providing more transparency and affect the perception of **responsibility** for impacts. Territorial-based approaches may help best in understanding urban planning and development needs, supply chain approaches may help to identify the role of the process chain and include regional infrastructure planning issues, whereas consumption-based approaches may reveal policy needs for behavioral and macro-economic changes (38). The ongoing trend towards consumption-based approaches reflects the need to discuss responsibility and fair resource usage (115). A complementary use of the approaches is warranted.
- A fundamental problem for all approaches is the definition of the **urban system's boundary** to use in the accounting. A universally accepted definition of what is 'urban' is not practical as cities in different countries exist in very different contexts. However, there is a need to delve deeper into the consequences of considering different boundaries (e.g. administrative vs. land-use) when carrying out research.
- There is a gap between research and practice that is greater in the UM field than in the energy use/GHG accounting field: thousands of communities have at least some experience in GHG inventorying, whereas very few cities have been involved in UM research. The availability of reliable **data and standard protocols** (such as GPC) is greater in the GHG accounting field and continues to grow rapidly. This is likely a reflection of the greater interest and momentum that urban responses to climate change currently have on the policy agenda, in contrast to broader resource use aspects. Mutual learning between the carbon inventorying field and UM field is desirable (19): on the one hand, many GHG accounting tools and initiatives could benefit from incorporating the impact of materials use into their methods. On the other, UM can learn from the application of consumption-based approaches in the GHG emissions field, and UM indicators would benefit from awareness initiatives that facilitate their communication to policy makers.
- Data collection involves costs and institutional requirements that are unknown or poorly researched in this area. Financially, the setting up of data collection systems by beneficiary cities should be considered over a timeframe of decades (136). Additionally, cities would benefit from joining national and international efforts to further develop databases usable at city scale,

including subnational multi-region input-output tables that resolve to finer geographical scales (17, 61).

- In both GHG accounting and the urban metabolism field we recognize a **dominance of (existing) published research on large global metropolises**, rather than on mid-size or small cities, which is where most urban growth is expected over the next decades. Moreover, studies that go beyond a limited number of city case studies are rare and international comparative approaches are almost non-existent.
- It is important to underline that quantifying the GHG emissions or resource flows around cities does not equate to measuring the **impacts** exerted by these flows on the environment. In fact there are a number of fields that are yet to be well integrated into the UM framework, such as resource depletion, ecosystem damage, impacts on biodiversity, or the link between urbanization and land-use change (137). To an extent, the use of LCA frameworks to complement these methodologies would allow this connection to be made, but the combined use of these methodologies is currently only in its early stages.

In this review we have decided to concentrate on two fields of research that are inherently linked. We acknowledge limitations in this approach, as our classification omits certain key city sustainability assessment topics, e.g. the measurement of urban vulnerability to climate hazards and climate change, for which research efforts are currently gaining momentum. We have also left aside the field of social and economic wellbeing indicators that complete the sustainability triangle although, naturally, cities are also strongly driven by social and economic sustainability goals. An often overlooked issue is, for instance, the fact that poverty affects approximately one in seven urban residents worldwide (138).

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Optional elements

Summary Points list

1. Rapid urbanization has led to an emergence of urban sustainability assessment methods that can help practitioners to find solutions for policy development and city planning. These may help to prioritize environmental aspects, locations or sectors in which to take action and help to design policy solutions at different governance levels.
2. With regard to environmental impacts two interconnected fields of research can be observed: on the one hand, a dominant trend of literature on the accounting and allocation of GHG emissions and energy use to cities (often called carbon footprinting) and, on the other, a re-emergence of studies focusing on urban metabolism or, in other words, the material and energy stocks and flows through cities.
3. Both fields of research are inherently linked as they originate from a systems approach - the UM field takes the city ecosystem as the fundamental unit of analysis and much of city GHG accounting literature applies the same notion. They both can consider cities as either producers or consumers and both recognize that urban environmental impacts can be local, regional or global. The two fields also show considerable divergence, in particular regarding the degree of application of the existing knowledge on the ground.
4. Urban energy and GHG accounting began in many cities in the 1990s. The recent introduction of the Global Protocol for Community-Scale Greenhouse Gas Emissions (GPC) aims to overcome the challenge of the much-contested incoherent approaches between cities and is designed to replace earlier protocols. However, systematizing different approaches and methodologies remains a challenge, in addition to the practical problems of widespread implementation.
5. Urban metabolism has a longstanding history and has made a major contribution to methods for accounting for material and energy flows, providing a basis for optimization of the city “ecosystem”. However it has been limited by the lack of standardized methods and paucity of data. Due to the data intensity and complexity of this field, there are relatively fewer applications of the method than in the energy/GHG accounting field, and most studies lack repeated data collection over time, or limit themselves to the study of single flows.
6. Territorial-based approaches may help best in understanding urban and regional planning needs, supply-chain approaches may help to identify the role of the process chain, whereas consumption-based approaches may reveal policy needs for behavioral and macro-economic changes. A complementary use of all the approaches is warranted.

Future Issues list

1. While the data situation is improving rapidly in the climate and energy fields, comprehensive data for quantifying urban resource flows is are as yet rarely available.
2. In the maturing field of urban sustainability assessment, the importance of benchmarking is also growing. Further research is needed on comparative (systematic) analysis of cities along their material and energy flow profiles or environmental footprints
3. Recently, city carbon inventorying gained impetus due to the establishment of the Global Protocol for Community-Scale Greenhouse Gas Emissions (GPC). International consensus on methodologies for accounting of cross-boundary emissions is currently sought. For urban metabolism to become "mainstream", a similar global agreement on what parameters are needed for basic reporting is required.
4. One promising field emerging in the literature is that of the measurement of synergies and trade-offs between city sustainability goals. Another promising area is the coupling of material flow analysis and LCA (connecting mass balances with insights about the varying pressures that different flows might have on the environment).
5. There is a need to develop a better link between descriptive approaches (as reviewed here) with more analytical ones, such as systems modeling and scenario analyses, in order to enhance the policy relevance of the former and accuracy of the latter.

Sidebar:

[note to editors: please place near section 5 (conclusions). Do advise on the possible cross-link of this sidebar to the paper on measurement of co-benefits of climate change mitigation which should appear in same issue of ARER]

Title: *Measuring synergies in city sustainability*

One particular field emerging in the literature is that of the measurement of synergies and trade-offs between city sustainability goals. In general, there is a clear perception from cities that are active in climate change that there are ancillary benefits that may come about through their actions (69) – however, conceptual models and integrated approaches to quantifying these are largely lacking (139, 140). One well understood area is the link between GHG mitigation actions and air quality and human health improvements (141), but the use of these methods to inform policy makers at the local level is still rare. As an example of a quantitative approach to an ex-ante assessment of policy co-benefits, Viguié and Hallegatte (72) use a simple integrated city model to assess the impacts of three different urban policies for Paris: a greenbelt policy, a zoning policy to reduce flood risk and a transportation subsidy (representing climate change mitigation, adaptation and equity goals, respectively). When considered as part of a policy mix, the consequences of each policy on the indicators combined in a non-additive manner. In a future where the synergies and conflicts between policy goals in an urban context are increasingly studied - and not just in the context of climate change mitigation -, city sustainability assessment methods like those reviewed in this paper will be needed to provide a solid data basis.

Literature cited

1. United Nations. 2012. World urbanization prospects, the 2011 revision. The Population Division of the Department of Economic and Social Affairs of the United Nations
2. Batty M. 2011. When all the world's a city. *Environ Plann A*. 43(4):765–72
3. UN-Habitat. 2011. *Cities and climate change: global report on human settlements*. Nairobi, Kenya: UN-Habitat, Earthscan
4. Seto KC, Güneralp B, Hutyra LR. 2012. Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proc. Natl. Acad. Sci. U.S.A.* 109(40):16083–88
5. Seto KC, Sánchez-Rodríguez R, Fragkias M. 2010. The new geography of contemporary urbanization and the environment. *Annu Rev Env Resour*. 35(1):167–94
6. Seto KC, Fragkias M, Güneralp B, Reilly MK. 2011. A meta-analysis of global urban land expansion. *PLoS ONE*. 6(8):e23777
7. Glaeser E. 2011. Cities, productivity, and quality of life. *Science*. 333(6042):592–94
8. Satterthwaite D. 2007. *The Transition to a Predominantly Urban World and its Underpinnings*. London: IIED. 99
9. Satterthwaite D. 2009. The implications of population growth and urbanization for climate change. *Environ Urban*. 21(2):545–67
10. Dodman D. 2009. Blaming cities for climate change? an analysis of urban greenhouse gas emissions inventories. *Environ Urban*. 21(1):185–201
11. Weisz H, Steinberger JK. 2010. Reducing energy and material flows in cities. *Curr. Opin. Environ. Sustainability*. 2(3):185–92
12. Hoornweg D, Freire M. 2013. Main report. vol. 1 of building sustainability in an urbanizing world: a partnership report. *Urban development series, knowledge papers no. 17*. The World Bank, Washington DC, USA

13. Güneralp B, Seto KC. 2013. Futures of global urban expansion: uncertainties and implications for biodiversity conservation. *Environ. Res. Lett.* 8(1):014025

14. McDonald RI, Marcotullio PJ, Güneralp B. 2013. Urbanization and global trends in biodiversity and ecosystem services. In *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities*, ed T Elmqvist, M Fragkias, J Goodness, B Güneralp, PJ Marcotullio, et al., pp. 31–52. The Netherlands: Springer

15. Romero Lankao P, Qin H. 2011. Conceptualizing urban vulnerability to global climate and environmental change. *Curr. Opin. Environ. Sustainability*. 3(3):142–49

16. IPCC. 2014. Summary for policymakers. in: climate change 2014: impacts, adaptation, and vulnerability. contribution of working group ii to the fifth assessment report of the intergovernmental panel on climate change. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press

17. Baynes TM, Wiedmann T. 2012. General approaches for assessing urban environmental sustainability. *Curr. Opin. Environ. Sustainability*. 4(4):458–64

18. Lechtenböhmer S, Barthel C, Merten F, Schneider C, Schüwer D, Seifried D. 2010. Redesigning urban infrastructures for a low emission future, an overview of urban low carbon technologies. . 3(2):1–16

19. Ramaswami A, Chavez A, Chertow M. 2012. Carbon footprinting of cities and implications for analysis of urban material and energy flows. *J. Ind. Ecol.* 16(6):783–85

20. Kennedy C, Baker L, Dhakal S, Ramaswami A. 2012. Sustainable urban systems. *J. Ind. Ecol.* 16(6):775–79

21. Davis SJ, Caldeira K. 2010. Consumption-based accounting of co2 emissions. *Proc. Natl. Acad. Sci. U.S.A.* 107(12):5687–92

22. Turner D, Williams I, Kemp S, Wright L, Coello J, McMurtry E. 2012. Towards standardization in ghg quantification and reporting. *Carbon Manage.* 3(3):223–25

23. Hertwich EG, Peters GP. 2009. Carbon footprint of nations: a global, trade-linked analysis. *Environ. Sci. Technol.* 43(16):6414–20

24. Larsen HN, Hertwich EG. 2009. The case for consumption-based accounting of greenhouse gas emissions to promote local climate action. *Environ. Sci. Policy*. 12(7):791–98

25. EEA. 2013. European union co2 emissions: different accounting perspectives. *No. 20/2013*, European Environment Agency

26. C40, ICLEI, WRI, WB, UNEP, UN-Habitat. 2012. Global protocol for community-scale greenhouse gas emissions (gpc) pilot version 1.0 – may 2012. C40 Cities Climate Leadership Group and ICLEI Local Governments for Sustainability in collaboration with: World Resources Institute, World Bank, UNEP, and UN-HABITAT

27. Kennedy C, Hoornweg D. 2012. Mainstreaming urban metabolism. *J. Ind. Ecol.* 16(6):780–82

28. Sippel M. 2011. Urban ghg inventories, target setting and mitigation achievements: how german cities fail to outperform their country. *Greenhouse Gas Measurement and Management*. 1(1):55–63

29. D' Avignon A, Carloni FA, Rovere ELL, Dubeux CBS. 2010. Emission inventory: an urban public policy instrument and benchmark. *Energ. Policy*. 38(9):4838–47

30. Wright LA, Coello J, Kemp S, Williams I. 2011. Carbon footprinting for climate change management in cities. *Carbon Manage.* 2(1):49–60

31. NDRC. 2011. China's policies and actions for addressing climate change-the progress report

32. Bader N, Bleischwitz R. 2009. Comparative analysis of local ghg inventory tools. College of Europe and Veolia

33. Kennedy C, Steinberger J, Gasson B, Hansen Y, Hillman T, et al. 2009. Greenhouse gas emissions from global cities. *Environ. Sci. Technol.* 43(19):7297–7302

34. Hillman T, Ramaswami A. 2010. Greenhouse gas emission footprints and energy use benchmarks for eight u.s. cities. *Environ. Sci. Technol.* 44(6):1902–10

35. Ramaswami A, Hillman T, Janson B, Reiner M, Thomas G. 2008. A demand-centered, hybrid life-cycle methodology for city-scale greenhouse gas inventories. *Environ. Sci. Technol.* 42(17):6455–61

36. Chavez A, Ramaswami A. 2013. Articulating a trans-boundary infrastructure supply chain greenhouse gas emission footprint for cities: mathematical relationships and policy relevance. *Energ. Policy*. 54:376–84

37. Ramaswami A, Chavez A, Ewing-Thiel J, Reeve KE. 2011. Two approaches to greenhouse gas emissions foot-printing at the city scale. *Environ. Sci. Technol.* 45(10):4205–6

38. Baynes T, Lenzen M, Steinberger JK, Bai X. 2011. Comparison of household consumption and regional production approaches to assess urban energy use and implications for policy. *Energ. Policy*. 39(11):7298–7309

39. Minx J, Baiocchi G, Wiedmann T, Barrett J, Creutzig F, et al. 2013. Carbon footprints of cities and other human settlements in the uk. *Environ. Res. Lett.* 8(3):035039

40. Duren RM, Miller CE. 2012. Measuring the carbon emissions of megacities. *Nature Clim. Change*. 2(8):560–62

41. Kennedy C, Steinberger J, Gasson B, Hansen Y, Hillman T, et al. 2010. Methodology for inventorying greenhouse gas emissions from global cities. *Energ. Policy*. 38(9):4828–37

42. IPCC. 2006. 2006 ipcc guidelines for national greenhouse gas inventories. IGES, Japan

43. WRI/WBSCD. 2013. *Greenhouse Gas Protocol*. http://www.ghgprotocol.org/

44. Kennedy C, Demoullin S, Mohareb E. 2012. Cities reducing their greenhouse gas emissions. *Energ. Policy*. 49:774–77

45. Bader N, Bleischwitz R. 2009. Measuring urban greenhouse gas emissions: the challenge of comparability. *S.A.P.I.EN.S. Surveys and Perspectives Integrating Environment and Society*
46. Kaselofsky J, März S, Schüle R. 2013. Bottom-up evaluation of municipal energy and climate policy : more than an alternative to top-down approaches?
47. Harris PG, Chow ASY, Symons J. 2012. Greenhouse gas emissions from cities and regions: international implications revealed by hong kong. *Energ. Policy*. 44:416–24
48. Ramaswami A, Chavez A. 2013. What metrics best reflect the energy and carbon intensity of cities? insights from theory and modeling of 20 us cities. *Environ. Res. Lett.* 8(3):035011
49. Cai B. 2012. Research on greenhouse gas emissions inventory in the cities of china. *China Population Resources and Environment*. 22(01):21–27
50. Gu C, Yuan X. 2011. Urban greenhouse gas inventory and methods in china. *Urban Environment & Urban Ecology*, pp. 5–8
51. Dienst C, Schneider C, Xia C, Saurat M, Fischer T, Vallentin D. 2013. On track to become a low carbon future city? – first findings of the integrated status quo and trends assessment of the pilot city of wuxi in china. *Sustainability*. 5(8):3224–43
52. Cai B, Zhang L. 2014. Urban co2 emissions in china: spatial boundary and performance comparison. *Energy Policy*. 66:557–67
53. Ecospeed AG. 2013. *ECOSPEED*. http://www.ecospeed.ch/
54. Kennedy C, Ramaswami A, Carney S, Dhakal S. 2011. Greenhouse gas emission baselines for global cities and metropolitan regions. In *Cities and climate change: responding to an urgent agenda*. World Bank
55. Chavez A, Ramaswami A, Nath D, Guru R, Kumar E. 2012. Implementing trans-boundary infrastructure-based greenhouse gas accounting for delhi, india. *J. Ind. Ecol.* 16(6):814–28

56. Braschel N, Posch A. 2013. A review of system boundaries of ghg emission inventories in waste management. *J. Clean. Prod.* 44:30–38

57. Paloheimo E, Salmi O. 2013. Evaluating the carbon emissions of the low carbon city: a novel approach for consumer based allocation. *Cities*. 30:233–39

58. Heinonen J, Kyrö R, Junnila S. 2011. Dense downtown living more carbon intense due to higher consumption: a case study of helsinki. *Environ. Res. Lett.* 6(3):034034

59. Larsen HN, Hertwich EG. 2010. Identifying important characteristics of municipal carbon footprints. *Ecol. Econ.* 70(1):60–66

60. Heinonen J, Junnila S. 2011. A carbon consumption comparison of rural and urban lifestyles. *Sustainability*. 3(8):1234–49

61. Wiedmann T, Wilting HC, Lenzen M, Lutter S, Palm V. 2011. Quo vadis mrio? methodological, data and institutional requirements for multi-region input–output analysis. *Ecol. Econ.* 70(11):1937–45

62. Wiedmann T, Barrett J. 2010. A review of the ecological footprint indicator—perceptions and methods. *Sustainability*. 2(6):1645–93

63. Watson D, Acosta-Fernández J, Wittmer D, Gravgård Pedersen O. 2013. Environmental pressures from european consumption and production. a study in integrated environmental and economic analysis. *2/2013*, European Environment Agency EEA, Copenhagen

64. Lenzen M, Peters GM. 2010. How city dwellers affect their resource hinterland. *J. Ind. Ecol.* 14(1):73–90

65. Heinonen J, Junnila S. 2011. Case study on the carbon consumption of two metropolitan cities. *Int J Life Cycle Assess*. 16(6):569–79

66. Schulz NB. 2010. Delving into the carbon footprints of singapore—comparing direct and indirect greenhouse gas emissions of a small and open economic system. *Energ. Policy*. 38(9):4848–55

67. Heinonen J, Junnila S. 2011. Implications of urban structure on carbon consumption in metropolitan areas. *Environ. Res. Lett.* 6(1):014018

68. The British Standards Institution. 2012. *Draft PAS 2070:2012: Specification for the assessment of greenhouse gas emissions of a city by direct plus supply chain, and consumption-based approaches - Draft 2.0 for public consultation*

69. CDP, C40 group. 2013. Wealthier, healthier cities - how climate change action is giving us wealthier, healthier cities

70. Yu W, Pagani R, Huang L. 2012. Co2 emission inventories for chinese cities in highly urbanized areas compared with european cities. *Energ. Policy*. 47:298–308

71. Güneralp B, Seto KC. 2008. Environmental impacts of urban growth from an integrated dynamic perspective: a case study of shenzhen, south china. *Global Environmental Change*. 18(4):720–35

72. Viguié V, Hallegatte S. 2012. Trade-offs and synergies in urban climate policies. *Nature Clim. Change*. 2(5):334–37

73. Wolman A. 1965. The metabolism of cities. *Sci. Am.* 213:179–90

74. Kennedy C, Cuddihy J, Engel-Yan J. 2007. The changing metabolism of cities. *J. Ind. Ecol.* 11(2):43–59

75. Kennedy C, Pincetl S, Bunje P. 2011. The study of urban metabolism and its applications to urban planning and design. *Environ. Pollut.* 159(8–9):1965–73

76. Zhang Y. 2013. Urban metabolism: a review of research methodologies. *Environ. Pollut.* 178:463–73

77. Barles S. 2010. Society, energy and materials: the contribution of urban metabolism studies to sustainable urban development issues. *J. Environ. Plann. Man.* 53(4):439–55

78. Goldstein B, Birkved M, Quitzau M-B, Hauschild M. 2013. Quantification of urban metabolism through coupling with the life cycle assessment framework: concept development and case study. *Environ. Res. Lett.* 8(3):035024

79. Bringezu S, Sand I v. d., Schütz, Bleischwitz R, Moll S. 2009. Analysing global resource use of national and regional economies across various levels. In *Sustainable resource management - trends, visions and policies*, ed S Bringezu, R Bleischwitz, pp. 10–51. Greenleaf Publishing
80. Barles S. 2009. Urban metabolism of paris and its region. *J. Ind. Ecol.* 13(6):898–913
81. Hendriks C, Obernosterer R, Müller D, Kytzia S, Baccini P, Brunner PH. 2000. Material flow analysis: a tool to support environmental policy decision making. case-studies on the city of vienna and the swiss lowlands. *Local Env.* 5(3):311–28
82. Hammer M, Giljum S. 2006. Material flow analyses of the regions of hamburg, vienna and leipzig. *6*, Sustainable Europe Research Institute (SERI)
83. Warren-Rhodes K, Koenig A. 2001. Escalating trends in the urban metabolism of hong kong: 1971–1997. *AMBIO*. 30(7):429–38
84. Newcombe K, Kalma JD, Aston AR. 1978. The metabolism of a city: the case of hong kong. *AMBIO*. 7:3–15
85. Schulz NB. 2005. Contributions of material and energy flow accounting to urban ecosystem analysis: case study singapore. *UNU-IAS Working Paper No._136*, United Nations University - Institute of Advance Studies
86. Schulz NB. 2007. The direct material inputs into singapore's development. *J. Ind. Ecol.* 11(2):117–31
87. Niza S, Rosado L, Ferrão P. 2009. Urban metabolism - methodological advances in urban material flow accounting based on the lisbon case study. *J. Ind. Ecol.* 13(3):384–405
88. Sahely HR, Dudding S, Kennedy CA. 2003. Estimating the urban metabolism of canadian cities: greater toronto area case study. *Can. J. Civ. Eng.* 30(2):468–83
89. Thériault J, Laroche A-M. 2009. Evaluation of the urban hydrologic metabolism of the greater moncton region, new brunswick. *Can. Water Resour. J.* 34(3):255–68

90. Kenway S, Gregory A, McMahon J. 2011. Urban water mass balance analysis. *J. Ind. Ecol.* 15(5):693–706

91. Müller D. 2006. Stock dynamics for forecasting material flows—case study for housing in the netherlands. *Ecol. Econ.* 59(1):142–56

92. Bergsdal H, Brattebø H, Bohne RA, Müller DB. 2007. Dynamic material flow analysis for norway's dwelling stock. *Building Research & Information*. 35(5):557–70

93. Cai ZF, Zhang LX, Zhang B, Chen ZM. 2009. Emergy-based analysis of beijing–tianjin–tangshan region in china. *Commun. Nonlinear Sci.* 14(12):4319–31

94. Vega-Azamar RE, Glaus M, Hausler R, Oropeza-García NA, Romero-López R. 2013. An emergy analysis for urban environmental sustainability assessment, the island of montreal, canada. *Landscape Urban Plan.* 118:18–28

95. Zhang Y, Yang Z, Liu G, Yu X. 2011. Emergy analysis of the urban metabolism of beijing. *Ecol. Model.* 222(14):2377–84

96. Huang S-L, Hsu W-L. 2003. Materials flow analysis and emergy evaluation of taipei's urban construction. *Landscape Urban Plan.* 63(2):61–74

97. Huang S-L, Lee C-L, Chen C-W. 2006. Socioeconomic metabolism in taiwan: emergy synthesis versus material flow analysis. *Resour. Conserv. Recy.* 48(2):166–96

98. Yuan Z, Shi J, Wu H, Zhang L, Bi J. 2011. Understanding the anthropogenic phosphorus pathway with substance flow analysis at the city level. *J. Environ. Manage.* 92(8):2021–28

99. Forkes J. 2007. Nitrogen balance for the urban food metabolism of toronto, canada. *Resour. Conserv. Recy.* 52(1):74–94

100. Brunner PH, Rechberger H. 2004. *Practical handbook of material flow analysis*. Boca Raton, FL: CRC/Lewis

101. Eurostat. 2013. *“Material flow accounts” - Statistics Explained*. http://epp.eurostat.ec.europa.eu/statistics_explained/index.php/Material_flow_accounts#

102. OECD. 2008. Measuring material flows and resource productivity – synthesis report. *OECD Work Programme on Material Flows and Resource Productivity*. OECD

103. Billen G, Garnier J, Barles S. 2012. History of the urban environmental imprint: introduction to a multidisciplinary approach to the long-term relationships between western cities and their hinterland. *Reg. Environ. Change*. 12(2):249–53

104. Ramaswami A, Weible C, Main D, Heikkila T, Siddiki S, et al. 2012. A social-ecological-infrastructural systems framework for interdisciplinary study of sustainable city systems. *J. Ind. Ecol.* 16(6):801–13

105. Seto KC, Reenberg A, Boone CG, Fragkias M, Haase D, et al. 2012. Urban land teleconnections and sustainability. *Proc. Natl. Acad. Sci. U.S.A.*

106. Minx J, Creutzig F, Medinger V, Ziegler T, Owen A, Baiocchi G. 2011. Developing a pragmatic approach to assess urban metabolism in europe. a report to the european environment agency prepared by technische universität berlin and stockholm environment institute. *01/2011*, Technische Universität Berlin

107. Hubacek K, Guan D, Barrett J, Wiedmann T. 2009. Environmental implications of urbanization and lifestyle change in china: ecological and water footprints. *J. Clean. Prod.* 17(14):1241–48

108. Feng K, Hubacek K, Minx J, Siu YL, Chapagain A, et al. 2010. Spatially explicit analysis of water footprints in the uk. *Water*. 3(1):47–63

109. Environment Agency. 2013. *London’s ecological footprint*. http://www.environment-agency.gov.uk/research/library/publications/115654.aspx

110. Scotti M, Bondavalli C, Bodini A. 2009. Ecological footprint as a tool for local sustainability: the municipality of piacenza (italy) as a case study. *Environ. Impact Asses.* 29(1):39–50

111. Kissinger M, Sussman C, Moore J, Rees WE. 2013. Accounting for the ecological footprint of materials in consumer goods at the urban scale. *Sustainability*. 5(5):1960–73

112. McDonald GW, Patterson MG. 2004. Ecological footprints and interdependencies of new zealand regions. *Ecol. Econ.* 50(1–2):49–67

113. Moore J, Kissinger M, Rees WE. 2013. An urban metabolism and ecological footprint assessment of metro vancouver. *J. Environ. Manage.* 124:51–61

114. Wilson J, Tyedmers P, Grant J. 2013. Measuring environmental impact at the neighbourhood level. *J. Environ. Plann. Man.* 56(1):42–60

115. Wackernagel M, Kitzes J, Moran D, Goldfinger S, Thomas M. 2006. The ecological footprint of cities and regions: comparing resource availability with resource demand. *Environ Urban*. 18(1):103–12

116. Blomqvist L, Brook BW, Ellis EC, Kareiva PM, Nordhaus T, Shellenberger M. 2013. Does the shoe fit? real versus imagined ecological footprints. *PLoS Biol*. 11(11):e1001700

117. Chester M, Pincetl S, Allenby B. 2012. Avoiding unintended tradeoffs by integrating life-cycle impact assessment with urban metabolism. *Curr. Opin. Environ. Sustainability*. 4(4):451–57

118. Pincetl S, Bunje P, Holmes T. 2012. An expanded urban metabolism method: toward a systems approach for assessing urban energy processes and causes. *Landscape Urban Plan.* 107(3):193–202

119. Li D, Wang R. 2009. Hybrid emergy-lca (heml) based metabolic evaluation of urban residential areas: the case of beijing, china. *Ecol. Complex.* 6(4):484–93

120. Brunner PH. 2011. Urban mining a contribution to reindustrializing the city. *Journal of Industrial Ecology*. 15(3):339–41

121. Pauliuk S, Venkatesh G, Brattebø H, Müller DB. 2013. Exploring urban mines: pipe length and material stocks in urban water and wastewater networks. *Urban Water J.* 0(0):1–10

122. Gaffney A, Huang V, Maravilla K, Soubotin N. 2007. Hammarby sjostad. stockholm, sweden: a case study. *CP249*, Urban Design in Planning

123. Kennedy C, Pincetl S, Bunje P. 2012. Reply to "comment on 'the study of urban metabolism and its applications to urban planning and design' by kennedy et al. (2011)". *Environ. Pollut.* 167:186

124. BRIDGE project. 2013. www.bridge-fp7.eu

125. Sustainable Urban Metabolism for Europe (SUME). 2013. www.sume.at

126. SUME. 2011. *Planning resource-efficient cities. SUME Synthesis Report.*

127. Pinho P, Oliveira V, Cruz SS, Barbosa M. 2013. Metabolic impact assessment for urban planning. *J. Environ. Plann. Man.* 56(2):178–93

128. González A, Donnelly A, Jones M, Chrysoulakis N, Lopes M. 2013. A decision-support system for sustainable urban metabolism in europe. *Environ. Impact Asses.* 38:109–19

129. Saldivar-Sali AND. 2010. *A global typology of cities: classification tree analysis of urban resource consumption*. PhD thesis. Massachusetts Institute of Technology. 140 pp.

130. Codoban N, Kennedy C. 2008. Metabolism of neighborhoods. *J. Urban Plan. D-ASCE*. 134(1):21–31

131. Castán Broto V, Allen A, Rapoport E. 2012. Interdisciplinary perspectives on urban metabolism. *J. Ind. Ecol.* 16(6):851–61

132. Newman PW. 1999. Sustainability and cities: extending the metabolism model. *Landscape Urban Plan.* 44(4):219–26

133. Huang S-L, Chen Y-H, Kuo F-Y, Wang S-H. 2011. Emergy-based evaluation of peri-urban ecosystem services. *Ecol. Complex.* 8(1):38–50

134. UNEP. 2013. *City-Level Decoupling: Urban resource flows and the governance of infrastructure transitions. A Report of the Working Group on Cities of the International Resource Panel. Swilling M., Robinson B., Marvin S. and Hodson M.*

135. Global City Indicators Facility. 2013. http://www.cityindicators.org/

136. Brunner PH. 2007. Reshaping urban metabolism. *J. Ind. Ecol.* 11(2):11–13

137. Yu Y, Feng K, Hubacek K. 2013. Tele-connecting local consumption to global land use. *Global Environmental Change*. 23(5):1178–86

138. Mitlin D, Satterthwaite D. 2013. *Urban Poverty in the Global South: Scale and Nature*. Routledge

139. Amekudzi A. 2011. Placing carbon reduction in the context of sustainable development priorities: a global perspective. *Carbon Manage.* 2(4):413–23

140. Amekudzi AA, Ramaswami A, Chan E, Lam K, Hon Meng W, Zhu D. 2011. Contextualizing carbon reduction initiatives: how should carbon mitigation be addressed by various cities worldwide? *Carbon Manage.* 2(4):363–65

141. West JJ, Smith SJ, Silva RA, Naik V, Zhang Y, et al. 2013. Co-benefits of mitigating global greenhouse gas emissions for future air quality and human health. *Nature Clim. Change*. 3(10):885–89

142. Klimabündnis Österreich. 2013. *CO2 Grobbilanz*. http://co2rechner.klimabuendnis.at/Intro.aspx

143. *GRIP*. The Greenhouse Gas Regional Inventory Protocol. http://getagriponemissions.com/index-cycle.html

144. *Accociation Bilan Carbon*. 2013. http://www.associationbilancarbone.fr

145. Aarhus University. 2013. *CO2 Calculator*. http://envs.au.dk/en/knowledge/air/emissions/co2-calculator

Tables:

Table 1. General systems approaches to the quantification of urban sustainability

[note to editors: we still need to decide whether we replace Table 1 with the proposed "iconic" figure, based on the feedback from the reviewer]

		Field of application in urban sustainability assessment	
		Energy use and GHG emissions	**Material, energy and substance flows (Urban metabolism)**
General approach	**Territorial (Production-based)**	Use of fuels by households and industries and production of electricity within a city's boundary; associated GHG emissions.	Direct use of water by households and industries within a city's boundaries; direct outflow of wastewater.
	Supply-chain based	Energy use and GHG emissions from energy and infrastructure supply chains (e.g. electricity production or waste treatment plants) that are outside the city boundary.	Upstream (i.e. outside the city's boundaries) impacts of water supply to serve local demand.
	Consumption-based	GHG emissions released globally during the production of all goods and services consumed within the city.	Food, materials, water inflows or waste outflows attributable to final consumption in the city.

Table 2. Selected examples of support tools for local GHG inventories

		CO_2 Grobbilanz	**ECO_2-Region**	**GRIP**	**Bilan Carbone**	**CO_2 Calculator**
GHG measured		6 Kyoto GHGs	CO_2 or 6 Kyoto GHGs[b]	6 Kyoto GHGs	6 Kyoto GHGs[c]	CO_2, CH_4, N_2O
Sector and scope (1 to 3) [f]	**Energy (Stationary)**	1+2	1+2	1+2	1+2	1
	Energy (Mobile)	3	1+2	1+2	1+2	1
	Waste	1+3	(-)	1+3	1+3	1
	AFOLU[d]	1+3	(-)	1+3	1+3	1
	IPPU[e]	(-)	(-)	1	1	(-)
Initiatives / cities using the inventory		Small and large communities in Austria	Climate Alliance, European Energy Award; > 800 cities and districts in Germany, Austria, Switzerland, Italy	31 metropolitan areas in Europe, USA and China	Nearly 6,000 cities (>50,000 inhabitants) and companies in France; introduced in Germany in 2013	Mainly Danish municipalities
Reference: **Country of Origin:**		(142) Austria	(53) Switzerland	(143) UK	(144) France	(145) Denmark

[a] The six Kyoto Protocol GHGs are: carbon dioxide (CO_2); methane (CH_4), nitrous oxide (N_2O); hydrofluorocarbon (HFCs); perfluorocarbon (PFCs); sulfur hexafluoride (SF_6)

[b] Depending on version

[c] Plus other direct GHGs such as CFCs

[d] Agriculture Forestry and Land Use Change

[e] Industrial Processes and Product use

[f] See Table 3

Table 3. Definitions of scopes under the Global Protocol for Community-Scale Greenhouse Gas Emissions (26)

Scope 1	All direct emissions from sources within the geopolitical boundary of the community
Scope 2	Energy-related indirect emissions that occur outside the community boundary as a consequence of consumption/use of grid-supplied electricity, heating and/or cooling within the community boundary.
Scope 3	All other indirect emissions that occur outside the boundary as a result of activities within the community's geopolitical boundary, as well as trans-boundary emissions due to exchange/use/consumption of goods and services.

Table 4 Correspondence of urban GHG emission source accounting categories with methodological approaches

		Approach:	Territorial		Supply chain		Con-sumption
		GPC-scope[a]:	1		2	3	
		spatial coverage:	in-boundary[c]		cross- boundary		
Sector (Source class acc. to GPC)		methodology:	Inventory[d]		life cycle data (energy-related)	life cycle data Mixed[h]	EEIOT[i]
Energy	I	**Stationary units**					
Energy	I.1	Residential Buildings	X	1A4b[d]	X[e]		X[g]
Energy	I.2	Commercial/Institutional facilities	X	1A4a	X[e]		
Energy	I.3	Energy Generation	(x)	1A1	X[f]		
Energy	I.4	Industrial Energy Use	X	1A2,1A5 1A4c	X[e]		
Energy	I.5	Fugitive Emissions	X	1B			
Energy	II	**Mobile Units**					
Energy	II.1	On-Road Transportation	X	1A3b	X[e]	total travel by residents (& commuters)[h]	X[g]
Energy	II.2	Railways	X	1A3c	X[e]		
Energy	II.3	Water-Borne Navigation	X	1A3dii	X[e]		
Energy	II.4	Aviation	X	1A3aii	X[e]		
Energy	II.5	Off-Road	X	1A3eii			
Non-Energy	III	**Waste**					
Non-Energy	III.1	Solid Waste Disposal	X	4A		X	
Non-Energy	III.3	Biological Treatment	X	4B		X	
Non-Energy	III.4	Incineration and open burning	X	4C		X	
Non-Energy	III.5	Wastewater Treatment	X	4D		X	
Non-Energy	IV	**Industrial Processes and Product Use (IPPU)**					
Non-Energy	IV.1	Industrial Processes	X	2A,2B, 2C,2E			
Non-Energy	IV.2	Product Use	X	2D,2F, 2G,2H			
Non-Energy	V	**Agriculture, Forestry and Land Use (AFOLU)**					
Non-Energy	V.1	Direct emissions from AFOLU	X	3			
Indirect	VI	**Other Indirect emissions**					
Indirect	VI.1	All other scope 3 emissions from all					X

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		sources							
	VI.2	All transboundary scope 3 emissions due to exchange/consumption of goods and services							
		a) All transboundary scope 3 emissions due to imports of infrastructure goods[j]					X		
		b) All other consumption related scope 3 emissions							X

[a] see Table 3

[c] Jurisdictional or geographic territory

[d] Emissions category according to IPCC (42)

[e] Upstream GHG emissions from fuel consumption typically taken from life cycle studies;

[f] GHG emissions related to (net) imports of electricity and heat, plus upstream emissions for fuels;

[g] GHG emissions from consumption of goods and services other than energy and transport are listed below under indirect;

[h] Transport: either calculated using local or national data on per capita distance travelled or GHG emissions from transboundary travel; Waste: inventory method;

[i] Environmentally extended input-output analysis

[j] not accounted for under energy and waste (I - III) according to Ramaswami & Chavez (48)

x = emission source is accounted for in the methodological approach --- = not occurring

Source: own table based on C40 et al. (26)

Figures:

Figure 1. Approaches to accounting methods used to measure the environmental impacts of urban systems. The sectors within the city's territory (diagonal fields) provide goods and services that are either consumed locally (pink) or elsewhere (gray). The cross-boundary supply chains shown are examples, and their impacts may be associated with inflows (pink) and outflows (gray). (Source: authors' own, based on 21, 37, 42.) (GPC, Global Protocol for Community-Scale Greenhouse Gas Emissions; see Table 3.)

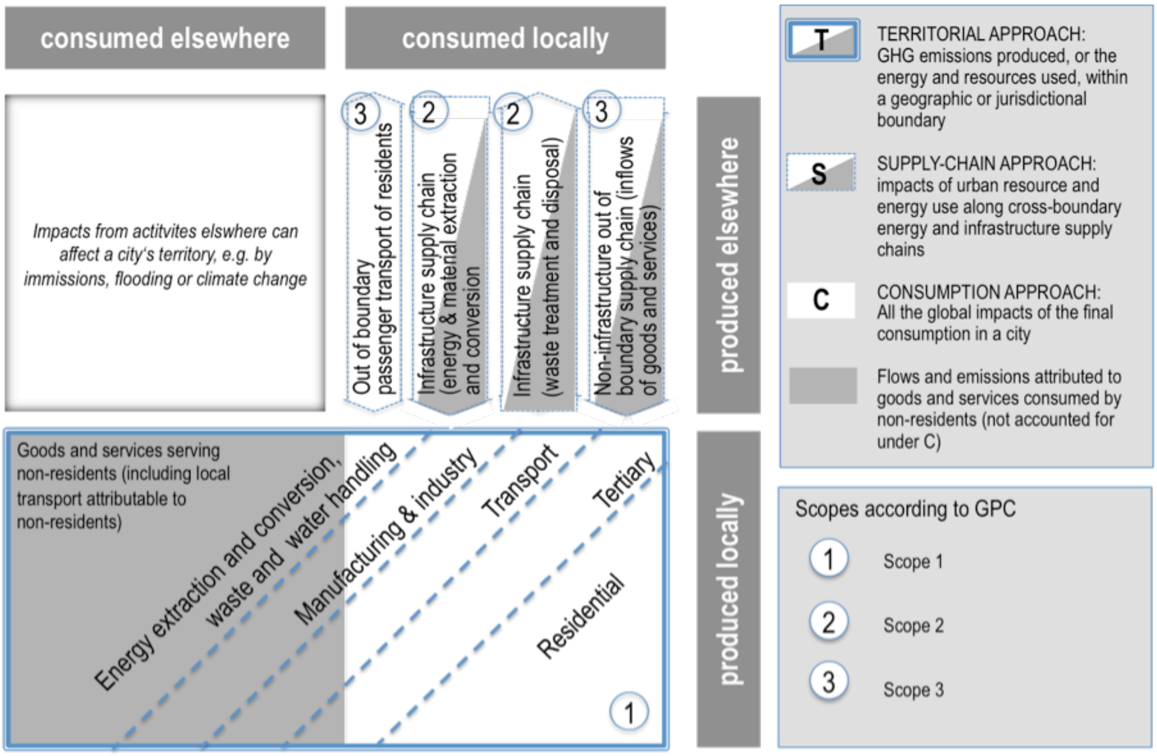

Figure 2. Simplified depiction of the urban metabolism system of stocks and flows through cities. Urban metabolism studies analyse the flows of materials, energy or of a particular substance within a well-defined system, connecting sources, metabolic pathways and sinks (based on: Castán Broto et al 2012; Barles, 2010, Kennedy, Pincetl, et al., 2011, Minx et al., 2011).

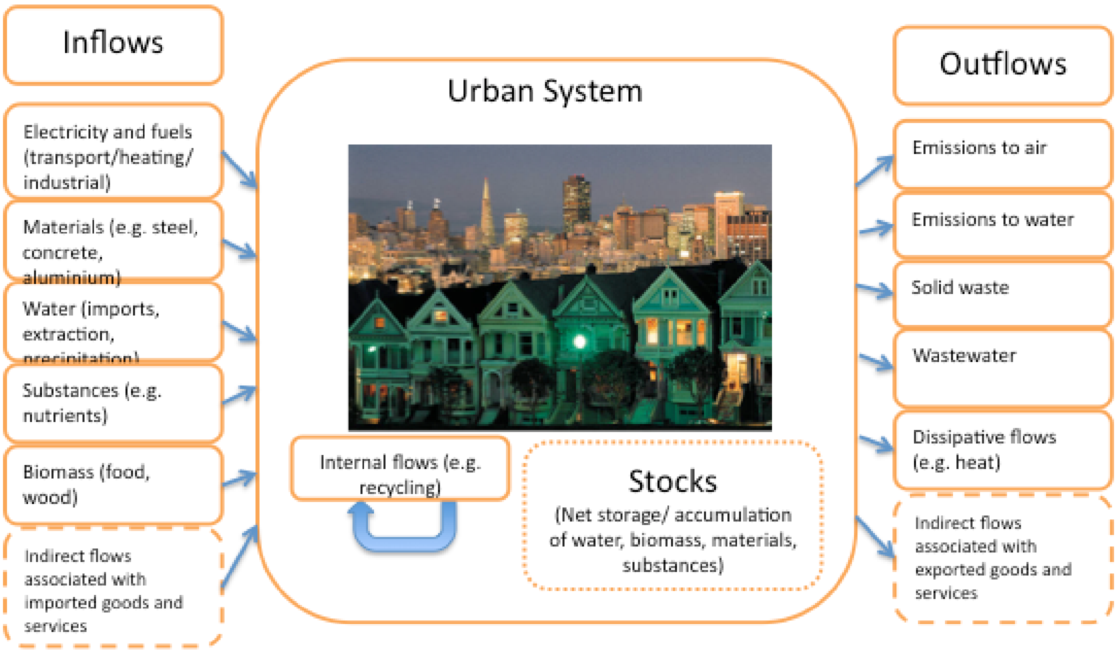

Figure 3. Examples of basic data requirements for urban metabolism studies, based on (Kennedy and Hoornweg, 2012; Minx et al., 2011)