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Urban metabolism: a review in the UK context

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Urban metabolism: a review in the UK context

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Contents

Abstract

This study discusses issues of urban metabolism and the implications for environmental sustainability. An urban metabolism can be thought of as the inflows of material and energy resources, the outflows of wastes and emissions and the retention of materials as stock in the built environment and infrastructure. To obtain a complete picture of the environmental and resource implications of the metabolism of a city, it is essential to assess these flows on a full life cycle basis, in other words to take account of effects that ripple up and down supply chains whether they occur within or outside the city boundary. This study summarises available studies on the metabolism of cities and includes examples of consumption accounts for cities in the UK and elsewhere. The material and energy flows depend on the structure and topography of a city; on the quality of its buildings and infrastructure; and on its social structure and the behaviour of its inhabitants. Historical trends and lessons from recent developments show how cities are changing and how their development can be guided. A key message is that cities are complex systems whose quality, efficiency and resilience cannot be addressed solely as economic issues.

1. The metabolism of cities: An introduction to the issues in describing, measuring and planning urban metabolism

Urban metabolism analysis takes account of all the processes that occur in cities, whether observed socially, economically or physically (Kennedy *et al*. 2007). The perspective in this paper is that of industrial ecology, (rather than e.g. political ecology) in which urban metabolism is studied in terms of the inflows of material and energy resources, the outflows of wastes and emissions, and the retention of materials as stock in the built environment and infrastructure. There are several practical reasons for studying urban metabolism. First, the quantities of energy, wastes, industrial production and other flows determined in urban metabolism are required when calculating the greenhouse gas inventories of cities. Second, measures of the efficiency of resource use in cities have been established based on urban metabolism studies (Oswald & Baccini, 2003; Zhang & Yang, 2007; Browne *et al*. 2009). Third, in an urban planning and design context, urban metabolism has also been used as a framework for undertaking the design of sustainable or low carbon districts/neighbourhoods within cities (as described in section 4.2). Fourth, urban retailers and consumers may make more informed purchasing decisions if they have knowledge of the impacts of their choices (Wiedmann 2012). Study of urban metabolism is thus an interdisciplinary endeavour and it has been undertaken by urban planners, engineers, political scientists, natural ecologists and industrial ecologists, amongst others (Castán Broto *et al*. 2012).

Further detail on urban metabolism is given in texts by Ferrão & Fernández (2013) and Baccini & Brunner (2012). The challenges of reducing energy and material flows in cities are discussed by Weisz & Steinberger (2010), while Calderon & Keirstead (2012) highlight the gap between accounting frameworks and policy tools. Literature reviews on urban metabolism have also been provided by Kennedy *et al*. (2011) and Zhang (2013). Loose parallels can be made between the metabolism of a city and the metabolism of an organism or an ecosystem; this metaphor is implicit in the term “industrial ecology” but is not essential and is perhaps irrelevant. The notion that cities are like organisms, or alternatively ecosystems, has been discussed and critiqued (Golubiewski, 2012; Kennedy, 2012b; Bettencourt, 2013). At some point the metaphor breaks down. Nonetheless cities and ecosystems are both types of complex systems that rely on their external environments for inputs of resources and for assimilation of wastes and create ordered structures at the expense of increasing disorder, i.e., environmental disruption outside of their boundaries.

The aim of this paper is to provide an exegesis and summary of the links between a city’s environmental impacts and its resilience and the well-being of its citizens. The paper is structured as follows. In this introduction, we set out the foundations of how we define and measure urban metabolism. In Section 2 we consider infrastructure of the built environment, looking at physical capital and its effect on urban metabolism. In Section 3 we discuss issues concerning consumption and management of waste, including behaviour change. Section 4 reviews historical changes and asks what lessons these past transitions might offer for the future of UK cities. The paper concludes with recommendations for how cities may be developed in ways that protect the environment and provide well-being both for cities’ inhabitants and those affected by cities’ activities.

1.1 Scales and system boundaries

When assessing the material stock of a city and flows of energy and materials into and out of it, i.e. analysing a city as a system, clear definition of boundaries is essential. This matter is not, however, straightforward, as there is flexibility in how an analyst defines the boundaries. (See Grubler *et al*. 2012, for approaches to analysing urban energy metabolism.) In this paper we are primarily concerned with spatial boundaries that include entire cities or urban regions. Often, for the practicalities of data collection, the system boundary will follow political boundaries of a city or metropolitan area. It is also possible to define sub-system boundaries within the overall urban system, or analyse smaller elements of cities, e.g., through the study of neighbourhood metabolism (Codoban & Kennedy, 2008).

A typical system boundary for analysing urban metabolism is shown in Figure 1-1. In this case the spatial boundary encapsulates the city or metropolitan region along with surrounding hinterlands, the air-shed above and the subsurface below. Thus the stocks and flows associated with the system, described in Section 1.2 below, can include both anthropogenic and natural components.

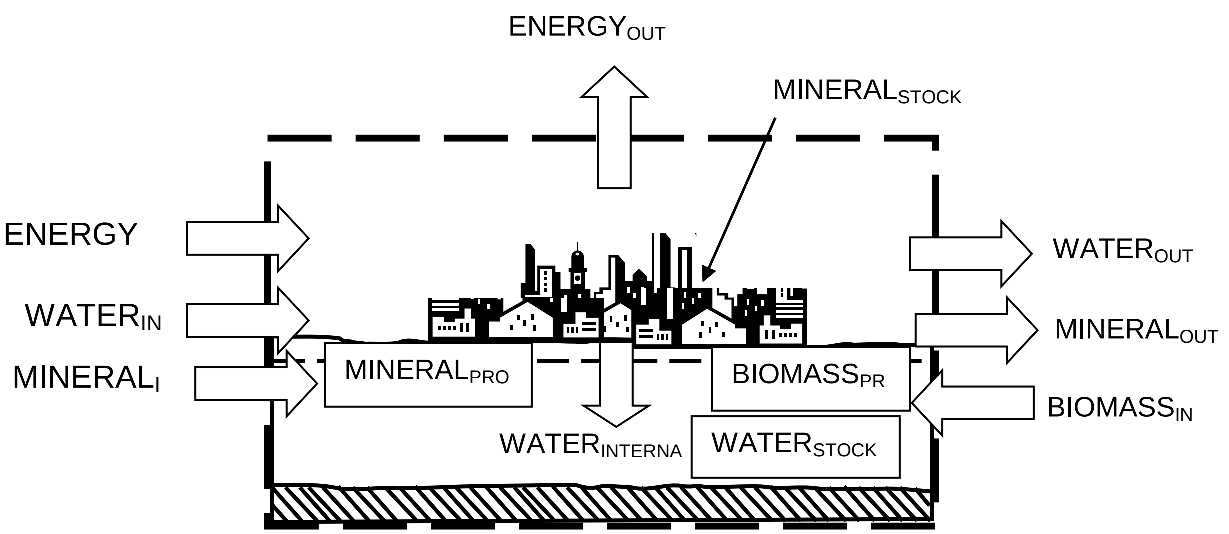

Figure 1-1. Urban system boundary showing inflows, outflows, internal flows, storage and production of biomass, minerals, water, and energy (adapted from Kennedy & Hoornweg, 2012)

When considering urban metabolism in order to understand and assess environmental sustainability it is also important to consider what 'perspective' is taken. Complementary to the city-wide perspective discussed above, a household consumption based approach places the system boundary around residential premises in a city and analyses the environmental impacts associated with flows to the households whether they originate inside or outside the city (Figure 1-2 a). It thus captures economy-wide impacts of consumption and lifestyles (Baynes & Wiedmann 2012) whereas the urban metabolism approach may be described as a hybrid production-consumption approach, including impacts both from production in the city and associated with net imports of energy and materials, often in a bulk form (Figure 1-2b). Both these approaches are relevant to assessing urban metabolism, and, for example, they are both specified for use in the Publicly Available Specification PAS 2070 '*Specification for the assessment of greenhouse gas emissions of a city – Direct plus supply chain and consumption-*

based methodologies'. A further discussion of the issue of perspectives, the two approaches discussed here and in PAS 2070 is provided in Section 1.3.1.

In addition to the two perspectives discussed above, a third group of approaches comes under the broad description of complex systems approaches, which attempt to capture the dynamics within cities at various scales (Figure 1-2c). They are not an amalgam of different perspectives, and no general way of defining spatial boundaries applies to this third category. Features of this third approach include representation of system interactions, feedbacks, and network relationships, sometimes using agent-based simulation (Baynes & Wiedmann 2012).

In this paper we draw on all three approaches.

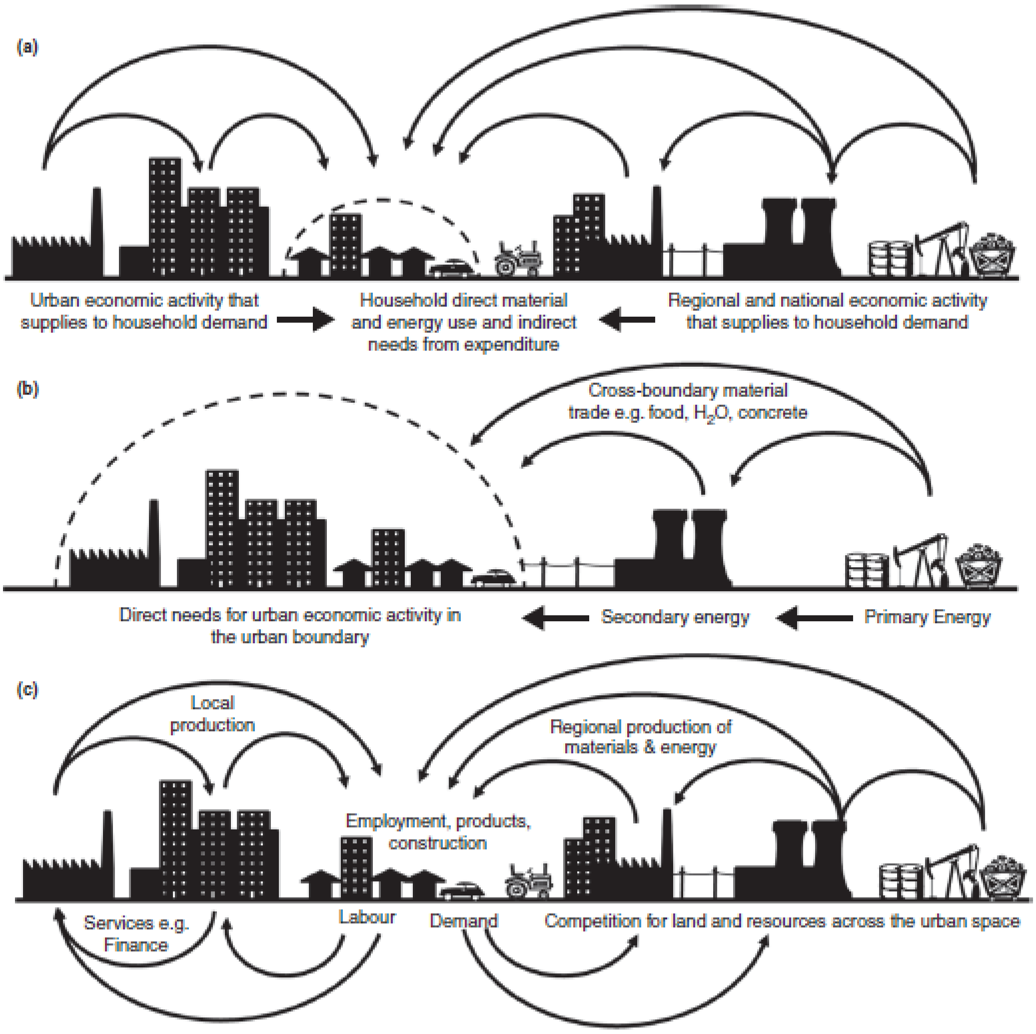

Figure 1-2. **Depiction of system boundaries and components in three generic approaches to assessing urban environmental sustainability, based on: a) household consumption; b) urban metabolism; c) complex systems (Figure 1 from Baynes and Wiedmann, 2012)**

1.2 Stocks and flows

The notion of stocks and flows is central to use of the urban metabolism as an accounting technique and as a framework for the analysis of urban systems. A *stock* can be defined as an asset at a particular point in time, whereas a *flow* represents transfers to or from a stock over a given time period. For example, the balance of a bank account at the start of the month (or any other moment in time) would represent a stock, whereas monthly payments and expenditures to and from this account would be flows. Flows can therefore change the level of a stock over time.

What constitutes a stock for urban metabolism analysis can be broadly defined. As a minimum requirement, one would expect to see measures of the economic capital of a city such as the materials embodied in buildings, infrastructure, equipment, and durable goods. However natural capital stocks, such as parkland, surface water or groundwater, may also be significant particularly when trying to understand a city's quality of life and resilience to natural hazards. Some significant flows may be directly relevant to these stocks. As a specific example, the rapid growth of Singapore since the 1960s can be clearly seen in its direct material consumption, with large imports of construction materials for the creation of new infrastructure (Schulz 2007). Other flows are more ephemeral. Electricity, natural gas, food, and water are often consumed very quickly and leave no obvious traces on major urban stocks. Nevertheless such flows are important for understanding the environmental impact of cities as well as the costs and vulnerabilities associated with maintaining a given standard of living.

A generic, comprehensive listing of the stocks and flows in the urban metabolism is shown in Table 1-1. The list, which is consistent with the system boundaries in Figure 1-1, includes all of the biophysical stocks and flows of the urban metabolism. The categories of stocks and flows were determined through an integration of the EURO-Stat system of material flow analysis, with measures of water, energy and substance flows that have typically been quantified in metabolism studies (Kennedy & Hoornweg, 2012). As the system boundary potentially includes some peri-urban areas, then the production of biomass (e.g. food, forestry) and minerals (mined) is included along with inflows, outflows and changes in stock.

Table 1-1. Aggregate inflows, outflows, stocks and production in the urban metabolism

Note: Square brackets indicate the appropriate units of measurement (mass in tonnes (t); energy in joules (J); electricity in kWh).

INFLOWS	OUTFLOWS
Biomass [t & J]	Waste Emissions [t]
food	gases
wood	solid
Fossil Fuel [t & J]	wastewater
transport	other liquids
heating/industrial	Heat [J]
Minerals [t]	Substances [t]
metals	Produced goods [t]
construction materials	
Electricity [kWh]	STOCKS
Natural energy [J]	Infrastructure / Buildings [t]
Water [t]	construction materials
Drinking (surface & groundwater)	metals
Precipitation	wood
Substances [t]	other materials
e.g. nutrients	Other (machinery, durable) [t]
Produced goods [t]	metals
	other materials
PRODUCTION	Substances [t]
Biomass [t & J]	
Minerals [t]	

Careful consideration has to be given to the practical amount and types of stock and flow data available and needed for an urban metabolism study. There is potentially a myriad of detailed data that could be collected for the urban metabolism, e.g. including inventory of individual goods or chemical elements. The appendix to a paper by Kennedy & Hoornweg (2012) proposes a practical list of broad urban metabolism measures determined by subjective judgement of the criticality of resource-related issues, while also considering data quality and recognising existing data collection practices for global cities. Another list of urban metabolism parameters for European cities is proposed by Minx *et al*. (2011). To collect even basic stock and flow data for UK cities may require the Office for National Statistics to adjust or develop new surveys, or for the cities themselves to become more proficient at data collection and management.

While there are several flows of particular importance in the urban metabolism, attention should also be given to monitoring stocks. Inflows of energy, water, food and basic materials such as cement and steel are of importance because of their connections to a variety of pressing environmental concerns from climate change to threats to global biodiversity. In the long-term, however, there are also critical stocks within the urban metabolism which also underlie the sustainability of cities, e.g. water stored in urban aquifers, toxic materials bound up in the building stock, and nutrients and precious metals in urban waste dumps (Kennedy *et al*. 2007). The energy stored in cities is also one measure of their resilience (discussed in Section 4.5).

There is a growing literature on the mathematical modelling of material stocks and flows in cities. Examples include: dynamics of metal stocks (Müller *et al*. 2014); spatial analysis of stock changes over time (Tanikawa & Hashimoto, 2009); and applications to urban water networks (e.g., Pauliuk *et al*. 2014, Venkatesh *et al*. 2014). Kennedy (2012a) presents a broad mathematical model of urban metabolism, showing how the quantity and performance of infrastructure stocks relates to most urban flows.

1.3 Resource use and impacts

1.3.1 Direct and indirect impacts

A 'life cycle' approach considers the whole supply system of a product from cradle (i.e. material or energy source) to grave (i.e. final disposal) (Baumann and Tillman, 2004); the life cycle approach is discussed further in Section 1.3.3. This perspective is essential to obtain a full assessment of the environmental impacts of a flow, particularly global impacts such as climate change due to greenhouse gas (GHG) emissions.

In the discussions which follow, it is important to distinguish between *direct* resource use, which is not assessed on a life cycle basis, and *indirect* (or *embedded* or *embodied*) resource use, which is assessed on a life cycle basis. Direct resource use and emissions occur within the geographical boundary of the area under consideration (a city in this case), whereas indirect resource use and associated emissions occur in the supply chain and therefore may arise within the urban area or anywhere else in the world. Although direct resource use (and associated emissions) can be useful when considering urban metabolisms, the life cycle approach gives a more complete picture. The distinction between direct and life cycle impacts is best explained through an example.

Let us consider a household that owns a car. The petrol used in the car can be recorded as the quantity of petrol used per year. This is its direct fuel use, and the emissions that arise when the petrol is burnt in its engine are direct emissions.

A whole life cycle approach would additionally take into account the *indirect* emissions and resource use. These include the infrastructure for extraction and processing of oil to produce the petrol used, and the transportation of the petrol to the garage where it is purchased. They also include the material resources – e.g. metals, plastic, rubber and glass – that went into manufacturing the car, and also the energy used in processing these materials and assembling and distributing the vehicle. A full life cycle approach also includes downstream effects associated with the disposal of the vehicle at the end of its service life, including credits for the use of materials and energy recovered from the scrap.

Building on this life cycle approach, in the next sub-section (1.3.2) we discuss ways of categorising resource use and associated emissions based on varying perspectives, then in Section 1.3.3 we discuss methods for assessing life cycle environmental impacts.

1.3.2 Approaches to assessing resource use and emissions

As mentioned in Section 1.1, different perspectives may be taken when estimating the material flows, energy use and associated emissions attributable to a city. Two fundamentally distinct perspectives are based on production and consumption (Druckman *et al*. 2008; Ferng 2003; Munksgaard & Pedersen 2001; Peters & Hertwich 2008a). The production perspective assesses flows on a territorial basis: it includes, for example, gas (and its associated emissions) used by households for cooking and heating, and transportation fuels (and the associated emissions) used within a city. It also includes energy use by companies in the city producing goods and services, regardless of the final destination of those goods and services. Assessing emissions from this perspective is relatively easy, with regional, sub-regional and local authority level data being produced by the respective administrative authorities[1].

In contrast, according to the consumption perspective, material flows, energy use and the associated emissions are attributed to final consumers, regardless of the location where the emissions arise. Therefore, for example, energy used within a city in the production of goods and services that are consumed elsewhere is excluded, whereas energy used outside a city in the production of goods and services consumed by households within the city is included. In other words, in the consumption perspective we take account of material flows, energy use, and the associated emissions embedded in imports into the city, but exclude those embedded in exports. Conversely, in the production perspective we take account of material flows, energy use, and the associated emissions embedded in exports and exclude those embedded in imports into the city. The consumption perspective is a great deal more complex and requires considerably more data than production accounting. At the city level, consumption accounting is most commonly carried out using Environmentally-Extended Input-Output (EEIO) Analysis (see Sections 1.3.3 and 3.1.1).

It is widely agreed that neither perspective alone is sufficient and that some combination of the two is required for effective policy-making (Bastianoni *et al*. 2004; Ferng 2003; Lenzen 2007; Lenzen *et al*. 2007; Munksgaard and Pedersen 2001). Conventional GHG reporting at the international scale and climate policies such as the Kyoto Treaty (UN 2004) follow the production perspective. This has the advantage that governments are able to apply effective regulatory and fiscal policies to limit energy use and emissions from operations within their own territories, whereas applying such measures to imported goods is more problematic. However, the consumption perspective is key for assessing the energy use and carbon emissions for which each nation must take responsibility (Bastianoni *et al*. 2004; Druckman *et al*. 2008; Munksgaard and Pedersen 2001; Peters and Hertwich 2006) because consumers are considered the ultimate drivers of all provision of goods and services.

The advantages and disadvantages of the two perspectives also apply at the city scale, although the distinction between the two approaches is not always made clear and explicit. Acknowledging that a combination of perspectives is required, PAS 2070 '*Specification for the assessment of greenhouse gas emissions of a city'* specifies two approaches (BSI 2013). Its aim is to provide a robust and transparent method for consistent, comparable and relevant quantification, attribution and reporting of city-scale GHG emissions, but the methodology and approach to defining system boundaries are equally applicable to the material, water and energy flows of a city and the associated emissions. PAS 2070 aims to encourage more holistic GHG assessments, greater disclosure and more meaningful benchmarking to help city decision-makers identify key emission sources and their drivers, the carbon dependence of their economy, and opportunities for more efficient urban supply chains. It is intended for use by organisations or individuals assessing environmental impacts of a city or an urban area, such as municipal or national governments, academic researchers, consultants, and others (BSI 2013).

[1] See for example: www.gov.uk/government/statistical-data-sets/sub-national-energy-consumption-statistics; http://data.gov.uk/dataset/electricity_consumption_at_local_authority_level

The two approaches specified by PAS 2070 are:

- Consumption-based using EEIO (see Sections 1.3.3 and 3.1.1); and
- Direct Plus Supply Chain (DPSC).

The DPSC method follows the GHG Gas Protocol for company reporting (WRI and WBCSD 2011), which categorises emissions as Scope 1, 2 and 3. These are defined as follows:

- Scope 1: Direct emissions from sources that are owned or controlled by the company (such as emissions from fuels combusted in a company's boilers);

- Scope 2: Indirect emissions from the generation of purchased energy (electricity, heat, steam or cooling) consumed by the company;

- Scope 3: Other indirect emissions that arise as a consequence of the activities of the company, but occur from sources not owned or controlled by the company. This is an optional reporting category in the GHG Protocol Corporate Accounting and Reporting Standard (WBCSD and WRI 2004).

  (See Figure 1 of Kennedy *et al*. (2010) for further explanation of how scopes 1, 2 and 3 apply to cities.)

The GHG Protocol and PAS 2070 both straddle the production and consumption perspectives. In PAS 2070, the DPSC methodology captures territorial GHG emissions and those associated with the largest supply chains serving cities, many of which are associated with city infrastructures. It covers direct GHG emissions from activities within the city boundary and indirect GHG emissions from the consumption of grid-supplied electricity, heating and/or cooling, transboundary travel and the supply chains of key goods and services produced outside the city boundary (e.g. water supply, food, building materials) (BSI 2013). (Note: there is potential for double-counting if DPSC based urban inventories are aggregated at a macro scale). Thus production perspective estimates form a subset of the flows assessed using DPSC methodology.

For examples of direct, indirect and hybrid methods of urban energy and greenhouse gas accounting see Baynes *et al*. (2011), Hillman and Ramaswami (2010), Ramaswami *et al*. (2011) and Chavez and Ramaswami (2013).

1.3.3 Life cycle impact assessment

The life cycle approach to evaluating the environmental and resource implications of the flows into and through a city (Figures 1-1 and 1-2) was introduced in Section 1.3.1. According to ISO 14044, life cycle assessment (LCA) (ISO 2006) is "a systematic tool for identifying and evaluating the environmental aspects of products and services from extraction of resource inputs to the eventual disposal of the product of its waste". Figure 1-3 illustrates this approach at the level of a single product.

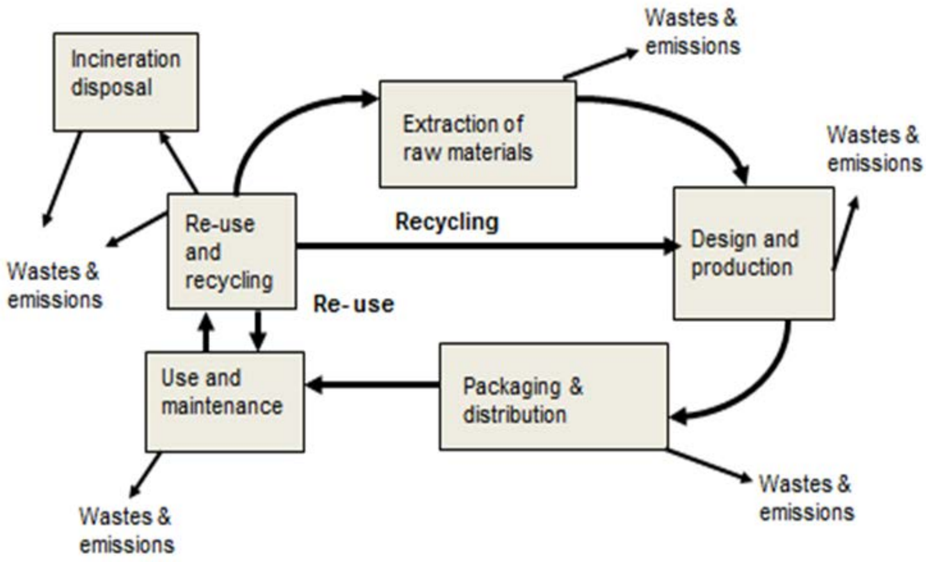

Figure 1-3. Product system with closed-loop; adapted from UNEP (1996)

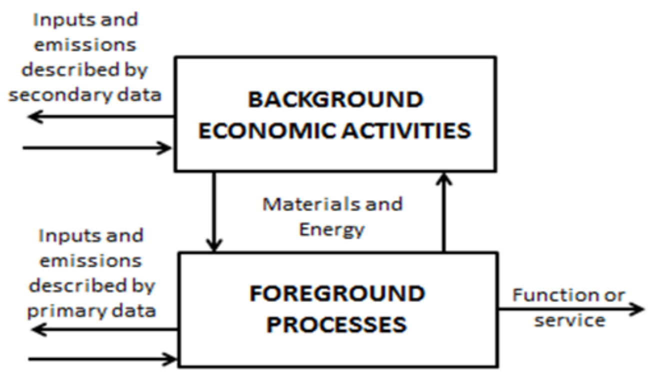

Figure 1-4. Background and Foreground (sub-) Systems

The methodology of LCA has developed so that the analysis conventionally passes through three phases:

1. Goal and scope definition, in which, inter alia, the boundaries of the product system are defined.

2. Inventory analysis, in which the materials and emissions entering and leaving the product system are identified and quantified.

3. Impact assessment, in which the inventory flows are translated into environmental impacts and primary resource usages.

To these phases may be added interpretation of the results. Due to the complexity and length of supply chains in the global economy, Life Cycle Assessment is notoriously data-intensive.

To confine the effort needed, it is common practice to distinguish at the inventory stage (see Figure 1-4) between:

- *Foreground*: the activities whose selection or mode or operation is controlled by the company or administrative body.

- *Background*: all other processes which interact with the Foreground, primarily by supplying or receiving material or energy.

In the context of urban metabolisms, the Foreground gives rise to the Scope 1 emissions while the Scope 2 and Scope 3 emissions arise from the Background system.

The inventory of the materials and emissions entering and leaving the system comprises a large body of data which is usually too detailed to reveal the most important environmental impacts. (In urban metabolism studies this data is useful in aggregate for comparing between cities). Phase (iii), Life Cycle Impact Assessment (LCIA), aims to evaluate and clarify the magnitude and significance of the potential impacts of the emissions from and inputs to the supply chain. Again, there are several different approaches (See Figure 1-5). The most established, and still most widely used, is the *mid-point* approach, which uses a set of recognised impacts each of which refers to a specific physico-chemical effect. Table 1-2 lists and briefly explains the most commonly used impact categories. Global climate change is the impact which usually dominates analysis of the environmental impacts of urban metabolisms; it represents the total contribution of all GHG emissions weighted according to their Greenhouse Warming Potential (GWP) relative to carbon dioxide over some specified period following emission, conventionally 100 years. Emissions identified in the inventory analysis can contribute to more than one impact category, as illustrated by Figure 1-6.

Table 1-2. Principal mid-point impact categories used in LCA

Global

Impact category	Description
Abiotic depletion potential	A measure of the potential of the system to contribute to the depletion of non-living natural resources that are not renewable within human lifetimes. The measure is based on remaining reserves and rates of extraction and expressed as equivalents to antimony (Sb).
Global warming potential (GWP 100)	The potential of a system to contribute to global warming, based on the radiative forcing capacities of atmospheric emissions. Expressed as equivalents to carbon dioxide.
Ozone layer depletion potential (ODP)	The potential of a system to contribute to the loss of stratospheric ozone, based on the change in the stratospheric ozone column due to gases emitted to atmosphere. Expressed as equivalents to CFC-11.

Regional and Local

Impact Category	Description
Acidification potential (AP)	The potential of the system to contribute to the acidification of freshwaters through emissions of acidifying substances to air. Expressed as equivalents to sulphur dioxide.
Eutrophication potential	The potential of the system to contribute to the eutrophication of freshwater through release of primary nutrients to water. Expressed as equivalents to phosphate.
Human toxicity potential	The potential for a system to impact on human health as a result of toxic substances released to the environment. It is the product of a number of factors covering the transfer, intake, fate and effect of different substances. Expressed as equivalents to 1,4-dichlorobenzene.
Freshwater aquatic ecotoxicity potential	The potential for a system to impact on the freshwater environment as a result of toxic substances released to the environment. It is the product of a number of factors covering the transfer, intake, fate and effect of different substances. Expressed as equivalents to 1,4-dichlorobenzene.
Marine aquatic ecotoxicity potential	The potential of a system to impact on the marine environment as a result of toxic substances released to the environment, expressed as equivalents to 1,4-dichlorobenzene.
Terrestrial toxicity potential	The potential for a system to impact on the soil environment as a result of toxic substances released. It is the product of a number of factors covering the transfer, intake and effect of different substances. Expressed as equivalents to 1,4-dchlorobenzene.
Photochemical oxidation potential (POCP)	The potential of a system to create tropospheric ozone through emissions of volatile organic compounds , expressed in equivalents to ethene.

The alternative approach uses *end-points*, which move further along the effect chain to estimate impacts on areas of social and economic concern (See Figure 1-5). Proponents of the end-point approach argue that expressing results in these terms – quantified impact on human health, for example – makes them more accessible to policy-makers and the lay public. Set against this, there is the unavoidable problem that proceeding from mid-points to end-points introduces further uncertainties into an assessment which is already subject to wide uncertainty even at the mid-point level. For this reason, the mid-point approach is still more widely used, including in this report.

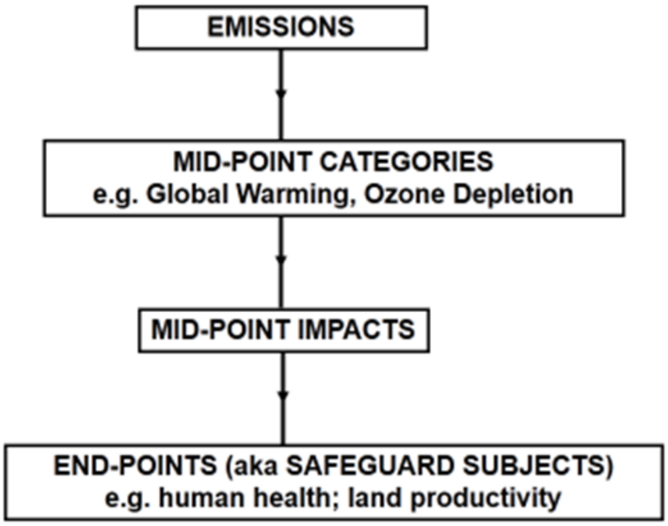

Figure 1-5. Life Cycle Impact Assessment: Mid-point and end-point impacts

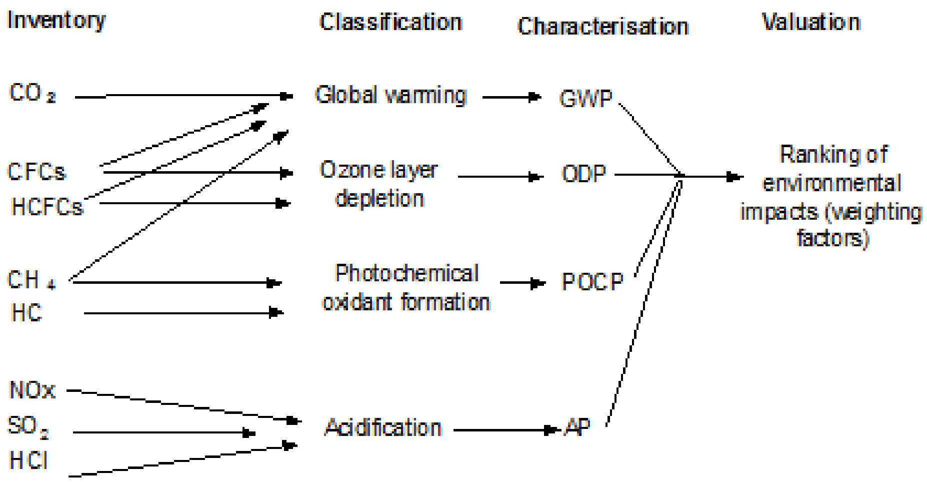

Figure 1-6. Life Cycle Impact Assessment using mid-point impacts

Approaches to presenting LCA results differ between different studies and practitioners. A common approach to interpreting mid-point impact estimates is to express them as fractions of reference values, such as the total contribution to each impact category by all the economic activities in the relevant country or economic area. This approach, known as *normalisation*, can help to show the significance of the different impacts and reveal which are greatest. However, appropriate reference values for normalisation are a matter of debate. Other approaches are more contentious. *Grouping* entails sorting and possibly ranking the (normalised) impacts in order of importance. *Weighting* aims to attach to each impact a factor indicating its importance, opening the possibility of aggregating the different impacts to a single score. However, this approach is also much criticised because it introduces another level of uncertainty and possibly subjectivity. The single score is sometimes expressed in terms of externalities (i.e. monetised damage costs), in which case the process may be called *Valuation*; this leads to a form of Cost-Benefit Analysis. Alternative aggregated impact metrics include the *Material Intensity per unit of Service* (MIPS) which sums the total mass of material moved or transformed to provide the product or service; this approach, developed at the Wuppertal Institute, is used mainly in Germany. Alternatively, the impacts may be aggregated as a conceptual area of land, known as the *Ecological Footprint* (EF) (Wackernagel & Rees 1996; McGranahan *et al*. 2005). In the context of urban metabolisms, the EF is intended to indicate the distributed hinterland on which the city draws. The concept behind the EF (Wackernagel & Rees 1996) is that “every individual, process, activity, and region has an impact on the earth, via resource use, generation of waste and use of services provided by nature” (Bergh & Verbruggen 1999: page 63). Impacts are accounted for by converting them to biological productive area of land required to provide the services, and the EF is the land area required to sustain the population in question. This is compared to the actual area of available land the population inhabits, and the degree of (un-)sustainability is calculated as the ratio of required land to available land (Lenzen & Murray 2003). Arguably, the EF has value as a heuristic for public communication, as in the concept of “One Planet Living” (Costanza 2000; Simmons *et al*. 2000). However, it has at best limited value as a quantitative indicator due to its lack of scientific robustness (van den Bergh & Grazi, 2010; Wiedmann & Barrett, 2010), combined with concerns regarding transparency and meaningfulness (HM Government 2005). The Urban Systems chapter of the Global Energy Assessment noted that “The ecological footprint of an urban area is invariably larger than the surface area of the city itself, which leads to the facile (but erroneous) conclusion that the city is ‘unsustainable.’” (Grubler *et al*. 2012). Cities are inherently concentrations of non-land based activities and have footprints beyond their boundaries. EFs continue to be used at the city scale (see Section 3.2) and attempts to improve the methodology have been made (e.g. Kitzes *et al*. 2009), but there is no realistic prospect that the EF can provide a sufficiently meaningful metric to be used in measuring performance or planning for increasing sustainability and resilience in urban metabolisms.

1.4 Construction and infrastructure: the built environment

Figure 1-7 summarises different perspectives on the notion of stocks and flows introduced in Section 1.2. In the conventional “linear” economy, material flows of food and goods become “waste” after they are consumed. Provided that the waste is not dispersed into the environment, for example if it is “stored” in a land-fill, it remains within the economy as a stock which could in principle be re-mined and exploited at some future date. Alternatively, the waste may be re-used more-or-less immediately within the economy; this is the concept behind the “closed-loop” or “circular” economy. If the environmental impact or “footprint” of a city is to be reduced, either materials and goods must be supplied with lower environmental impact by improving their life cycles (Section 1.3.2.), or the flows must be reduced by better or multiple use within the city. The “closed-loop” economy aims to do the latter.

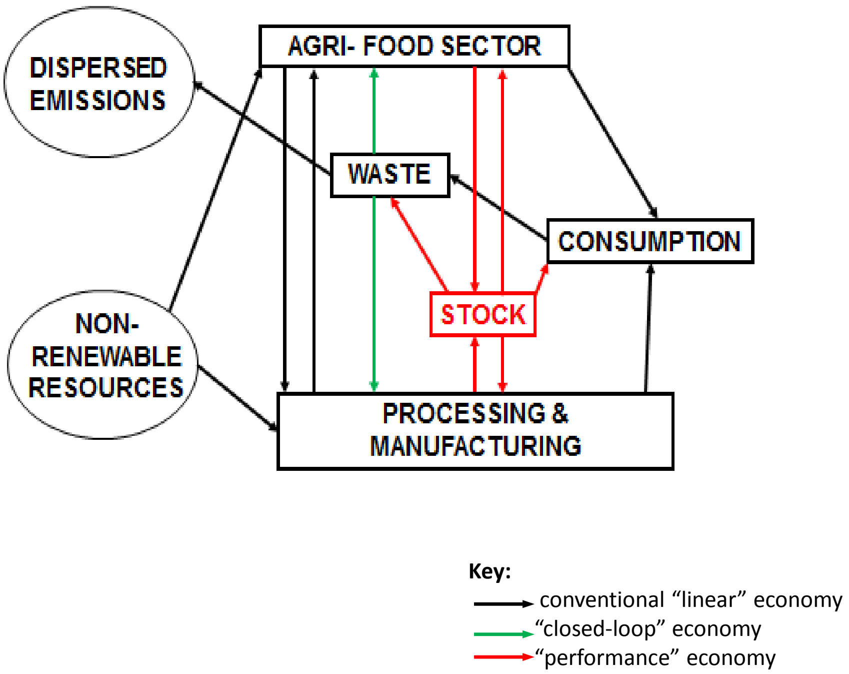

Figure 1-7. Flows and stocks in the Economy

However, in the “performance” economy model, advanced most notably by Stahel (2010), the focus is placed on the stocks in the economy: i.e. the built environment, material infrastructure and manufactured products such as vehicles and appliances. In this view, flows other than goods (primarily food) that are immediately consumed are related to building up and maintaining the stock. This leads to emphasising the importance of product and infrastructure durability as a way of reducing a city’s “footprint” by improving its resource efficiency and also improving the quality of life; this is explored further in Section 2.

1.5 Consumption of materials and energy

As already discussed, satisfying the social and economic functions of a city requires the consumption of substantial amounts of material and energy. These flows give rise to a range of environmental impacts, both locally and globally, and also at the current time plus at more distant time horizons. This section examines our understanding of how urban consumption activities affect these resource flows. We start by considering how some goods are purchased to meet demands that could be satisfied by alternate means, and the opportunities this might present for reducing resource flows and their associated impacts (Section 1.5.1). In Section 1.5.2 we consider changes in consumption patterns, and in Section 1.5.3 we consider issues of inequalities in consumption. The section ends with a brief look at issues arising due to bunker fuels for transportation, which can be problematic when assessing the city metabolism.

1.5.1 Consumption goods versus derived demands

In this sub-section we focus on the motivation behind resource consumption, which leads to a distinction between *consumption goods* and *derived demands* (Hirschey 2009).

Consumption goods represent the direct satisfaction of individual needs or wants. So for example, a vehicle might be purchased by a household because they get pleasure from owning a vehicle *per se*. In contrast, the fuel to run that vehicle can be considered as a *derived demand.* Derived demands occur as a result of demand for other goods or services, in this case mobility. So for a consumer who needs or wants a mobility service, the use of their vehicle to satisfy this consumption implies a derived demand for the fuel (petrol, diesel, electricity, etc.). Derived demands represent a unique opportunity for increased efficiency as new technologies or business models can open up alternative ways of providing the same end service but with a reduced derived demand. A good example would be public transport, which also provides the mobility service but with much less energy consumption per passenger. Similarly Sivaraman *et al*. (2008) showed that, by switching from a traditional bricks-and-mortar video rental service to one in which the DVDs are posted directly to consumers reduced the primary energy consumption required for the service of watching a film by nearly one-third. New e-commerce models, for example online streaming of video, could offer further savings in energy and material consumption but still provide consumers with the same (or comparable) end service. A word of warning is due here: although significant savings are possible in many cases, savings are often not as large as expected due to the rebound effect (Sorrell 2007: Boulanger *et al*. 2013; Brännlund *et al*. 2007; Chitnis *et al*. 2013; Druckman *et al*. 2011; Maxwell *et al*. 2011; Murray 2012; Saunders 2013; Turner 2009).

1.5.2 Changing patterns of consumption

Consumption by households is a complex phenomenon and the challenge of influencing consumption patterns is one of the major problems facing policy-makers in relation to sustainable development. More is said below in section 3.4 on theories of consumer behaviour and approaches to changing consumption patterns. Lifestyles can be seen as the overall patterns of everyday consumption choices and habitual practices, and they reflect different modes of consumption. These include: consumption reflecting personal identity, imitation of peers, shifts in fashion and technology; consumption shifts occasioned by new product standards and innovations in systems of provision; consumption organised around specific life events and transitions (house moves, birth of children, etc.); and the everyday ‘invisible’ and habitual practices of ordinary consumption organised around subsistence, work routines and household maintenance (Gronow and Warde 2001; Shove *et al*. 2012; Shove and Spurling 2013).

Changing patterns of consumption in urban environments can, in particular, be due to changes in numbers of residents and the socio-demographic mix, and changes in expenditure patterns.

Changes in the numbers of residents of cities is covered in a sister Foresight Working Paper to this paper: “People in Cities: The Numbers” by Tony Champion and therefore will not be repeated here. Suffice to say that some cities such as London, Milton Keynes and Bristol have increased in population between 1981 and 2011 whereas others, such as Newcastle, Glasgow and Liverpool have decreased (Champion Forthcoming). The age distribution of city residents has also changed over this time period, with an overall aging population, where the largest cities have the youngest age structure (Champion Forthcoming). Socio-demographics are covered in the sister Foresight Working Paper “Urban Social Disparities” by Philip Rees, Paul Norman and Helen Durham (Rees *et al*. Forthcoming).

Consumption patterns of a city can be affected by the socio-demographic mix as, for example, older consumers, or those of ethnic minorities, may tend to purchase different goods and services. This is because although some of the goods and services are purchased to meet subsistence needs such as food, clothing and shelter, goods also fulfil a more complex role in the “social conversations” that allow us to participate in the life of society (Douglas 2006; Douglas & Isherwood 1996; Jackson 2005; Jackson 2006; Jackson 2009). As expressed by Druckman and Jackson (2009: 1800): “We use goods to help to express our relationships with friends and family; to find ourselves a mate; to show which group we belong to; and to mark important occasions such as birthdays through, for example, the giving of gifts. Material goods are also implicated in the creation and maintenance of identity and the marking of status in society".

The strongest predictor of consumption of goods and services by residents of cities and the associated resource flows and emissions is income, with higher income households generally having higher flows (Buchs and Schnepf 2013; Gough 2013). However, there is a wide variation within each income group as the *mix* of goods and services purchased is key. Previous studies have shown that the highest environmental impacts are due to the broad categories of housing (including heating), food, and recreation (including transport) (Collins *et al*. 2006; Druckman & Jackson 2009; Nijdam *et al*. 2005; Peters & Hertwich 2006; Tukker 2006; Tukker & Jansen 2006; Vringer & Blok 1995; WWF-UK 2006). Chitnis *et al*. (2012) for example, carried out an econometric study which looked at past trends of total household expenditure classified into 16 categories in order to forecast future expenditures. The study found that household expenditure in almost all categories is predicted to rise. In the three future scenarios explored, the only exceptions to this were found to be 'alcoholic beverages and tobacco' and 'other fuels' expenditure, and 'electricity and gas' expenditure in the 'low' scenario only (Chitnis *et al*. 2012). The share of predicted expenditure is expected to change over time, with 'recreation and culture' taking over from 'other housing' as the highest expenditure category in 2020 and 2030 (Chitnis *et al*. 2012).

These estimates depend, of course, upon the assumptions adopted in the scenarios, but give a strong signal that consumer expenditure up to 2030 is expected to increase. Also, as explained in Section 1.3, the materials flows and energy used in production of goods and services purchased by UK city dwellers arise in both the UK and abroad. Due to the type of goods imported and tighter environmental standards in the UK than in some of our trading partner countries, the environmental impacts tend to be proportionately higher than if they were produced in the UK (Davis & Caldeira 2010; Druckman *et al*. 2008; Li and Hewitt 2008). For example, although generally less than 15% of household expenditure is on imported goods, analysis of the embedded GHG emissions shows that imported goods and services accounted for around 40% of total GHGs attributed to households in 2004 (Druckman and Jackson 2009). Thus unless policies are put in place to reduce the material and energy intensity of goods and services wherever they are produced, environmental impacts due to UK consumption are predicted to increase (CCC 2013; Davis & Caldeira 2010; Peters & Hertwich 2008b).

The clothing sector provides a good example of the increasing consumerist nature of society (Jackson 2009), in which we have witnessed trends in expenditure and that have resulted in increased material flows, energy use, and associated emissions. Due to the advent of the phenomenon known as 'fast fashion' or the 'Primark effect', i.e. the tendency to keep clothing for a shorter time before being disposed of (Morgan and Birtwistle 2009), increasing volumes of clothing are being produced and purchased. Flows of textiles through urban environments have therefore increased, with a marked increase in the quantity of textiles in municipal waste throughout the "Global North". Although there are attempts to encourage re-use and recycling (such as Marks and Spencer's Shwopping initiative[2]) much of the material is nevertheless disposed of in landfills.

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[2] http://help.marksandspencer.com/support/products-services/shwopping?rdUrl1=/FAQ-Shwop/b/1723541031

Arguably the most important effect of increased consumerism is the rising levels of carbon emissions that it induces. The Committee on Climate Change estimated that since 1993 the UK's carbon footprint increased by around 10%, despite increases in the efficiency of production in the UK (CCC 2013). The analysis shows that emissions assessed according to the consumption perspective are around 80% higher than those assessed according to the production perspective, reflecting that the UK has outsourced much of its GHG intensive production processes overseas (CCC 2013). Other effects concern resource scarcity, for example with regard to 'rare earth' elements which are essential for efficient electricity generation in wind turbines, and the platinum group metals which are essential for catalytic reduction of air pollutants (Erdmann & Graedel 2011).

1.5.3 Distribution issues

As mentioned in the previous section, the strongest predictor of consumption of goods and services by residents of cities and the associated resource flows and emissions is income, but there is a wide variation of expenditure patterns within each income group. So although urban metabolism studies often deal with aggregate figures, the distribution of consumption across a population can be of great policy interest, both in terms of reducing what might be termed 'excess consumption' by high income households and improving the quality of life for low income consumers.

High income households generally acquire larger volumes of material goods and use more energy than low income households (Buchs & Schnepf 2013; Gough 2013). However, because luxury items tend, in general, to be less material and energy intensive than necessities, as incomes rise expenditure tends to be less material and energy intensive (Tukker *et al*. 2010; Clift *et al*. 2013). Hence carbon emissions due to consumption (for example) are not directly proportional to income (Tukker *et al*. 2010).

Living in UK and other European cities is generally less energy and carbon intensive than rural living due to several factors (Buchs & Schnepf 2013; Tukker *et al*. 2010): first travel distances are likely to be shorter and public transport more available, thus energy use due to transportation tends to be less in cities than in rural areas; second, dwellings tend to be smaller and therefore require less energy to heat and light; third, due to higher density of housing and the heat island effect, energy use for heating also tends to be lower than in rural areas; fourth, urban dwellings generally have access to main supplies of gas whereas rural dwellings are more likely to rely on burning oil or LPG gas from cylinders for heating. These other fuels can be less carbon efficient for space heating (Defra 2012) and entail transportation for delivery of the fuel, and thus can be less environmentally efficient than centrally supplied gas. Finally, research into the effects of ethnicity has found that households where the head of the household is 'non-white' tend to have significantly lower embedded carbon dioxide emissions than their white counterparts (Buchs and Schnepf 2013; Tukker *et al*. 2010), and this is of relevance here as ethnic minorities are more likely to live in cities than rural areas. For example, according to Garner & Bhattacharyya (2011), London is home to disproportionate numbers of ethnic minority people and, generally, as a result of original settlement patterns around the larger industrial bases throughout the Midlands and the North of England, non-whites tend to live in urban areas across other parts of England too (Garner & Bhattacharyya 2011).

Other factors that affect the material and energy consumption of households include household size, where economies of scale mean that more people living under the same roof generally gain efficiencies of scale in energy use for household heating but not in terms of transportation

(Buchs & Schnepf 2013; Gough 2013). Diet is another factor (Garnett 2008; Rosa & Dietz 2012; Tukker *et al*. 2011).

Overconsumption or unnecessary consumption may occur due to inefficient pricing of a resource. A good example of this is water consumption. At present only about 45% of UK households have water meters installed, though in the Thames Water area the figure is only 30%, in part due to difficulties installing meters in flats (Thames Water 2013b). However evidence from a range of countries suggests that having consumers pay for what they use, rather than a fixed charge based on the size or occupancy of their property, can lead to reductions in water consumption of 1.4-56% (1.4—22% in the UK) (Inman & Jeffrey 2006).

On the other hand, increased resource charges may cause problems for low-income households. Fuel poverty is a particular challenge in the UK with approximately 2.7 million households in fuel poverty, a figure that is expected to rise slightly by 2016, with a significant increase in the fuel poverty "gap" (i.e. the financial cost of taking a household out of fuel poverty) (Hills 2012; DECC 2013).[3] Fuel poverty has significant impacts on health, and a conservative estimate suggests that it may be responsible for thousands of excess winter deaths each year (Hills 2012). While fuel poverty and general income poverty are clearly related, recent research has shown that "under-consumption" against needs is also an issue for the highest income households, where properties may be large and hard to heat, and also among single-occupant households (Hirsch *et al*. 2011).

These results indicate that a distributional analysis of resource consumption within a population is not simply a question of identifying low income households and improving their access to accepted levels of service. Other households – those with high-incomes, small children, illness, single-occupants – also have unique patterns of consumption that could be judged as being excessive or insufficient.

1.6 Bunker fuels

Where urban metabolism studies are conducted to support inventorying of GHG emissions for cities, then other flows may be recorded. An example of this is marine and aviation *bunker fuels* which are loaded onto ships and aircraft within urban regions. These contribute to scope 3 GHG emissions for cities (Kennedy at al. 2009, Ramaswami *et al*. 2008). However, in many cities, airports are outside city boundaries and this raises issues concerning the appropriateness of chosen boundaries depending on the purpose of the research being carried out. Also, there is a tendency for bunker fuels to be purchased in the cheapest location, and thus reporting of the fuel use and associated emissions attributable to each nation/city is problematic. Indeed, Gilbert *et al*. (2010) estimated that UK shipping emissions could be six times higher than official estimates. Thus it is clear that the uncertainties surrounding estimates of such resource flows and their associated emissions are considerable, and depend on the assumptions made in their allocation.

[3] Note that the government recently changed the definition of fuel poverty to focus on those households with both low-income and disproportionately high energy expenditures. See (Hills 2012).

2. Infrastructure and the built environment: physical capital and its effect on urban metabolism

It was pointed out in Section 1 that the built infrastructure of a city largely determines its metabolism and has a very strong effect on the quality of life of its citizens. The following section provides a brief summary of research and current thinking on the resource use and environmental impacts embodied in the built environment and the life cycle balance between investment in infrastructure and consumption of materials and energy, with transport as a specific and important example. The tension between increased investment in infrastructure and resultant reduction in consumption and the imperative to ensure that investments in infrastructure reduce consumption and enhance quality of life has been articulated, for example, by the Royal Commission on Environmental Pollution (RCEP 2007, pp.28-29):

"...the figures (on the embodied impacts of construction materials)…imply a contradiction between the government's desire to build more houses, and for instance its targets on CO_2 emissions and waste generation. In addition, the wider impact of the underpinning services and infrastructure required to support these new communities may further strain the environmental wellbeing of future city areas. If these tensions are to be resolved, the choices are:

- *action to reduce the environmental impacts of new and existing homes, their associated public buildings and places of work, and other necessary infrastructure;*
- *action in other sectors of the economy to offset the increased impact of the domestic sector, although this is likely to provide only limited room for manoeuvre given that other sectors face their own challenges;*
- *curtailing house building; or*
- *some combination of these strategies."*

The connection between material consumption and urbanisation is recognised but not developed in the 5th assessment report of the Intergovernmental Panel on Climate Change (IPCC 2014).

2.1 Quality of stocks and service life

A simple analysis to quantify the importance of quality of a stock such as the built infrastructure in a city is given in the Appendix, derived from an analysis by Clift & Allwood (2011) and based on a formulation introduced in the IPCC 5th Assessment Report (2014) which distinguishes between the energy use and emissions associated with providing the materials needed to maintain the stock, the intensity of use of the stock and its service life. Recognising that normal commercial pressures act to minimise energy use in industrial processes so that the scope for reducing emissions from primary production is limited (IPCC 2014) and that recycling usually requires less energy than primary production whilst re-use requires even less (Allwood *et al*. 2010), the priority order of options for reducing energy use associated with most components of urban infrastructure and material products is:

- Extend service life to reduce material demand;
- Intensify use to reduce material stock requirements;
- Increase the proportion of post-use product re-used;

- Increase the proportion of post-use product recycled;
- Reduce the energy required for recycling;
- Reduce the energy required for re-use;
- Reduce the energy required for primary material production.

The specific case of clothing (Section 1.5.2) provides an immediate illustration of the applicability of this analysis: short product life leads to high throughput. For the built environment, the analysis provides a basis for assessing the efficiency (or quality – see section 1.4) of buildings and infrastructure and also the benefit of retrofitting existing buildings and infrastructure (a form of re-use) and recycling components and materials from buildings demolished.

2.2 Construction materials and their embodied impacts

"Five key materials – steel, cement, plastic, paper and aluminium – dominate emissions from industry, and producing them accounts for 20% of all global emissions from energy use and industrial processes" (Allwood & Cullen 2012, p.18). Of these materials, steel and cement are the most significant for buildings and urban infrastructure. Increasing urbanisation, particularly in the "global south", will increase demand for steel and cement, a trend which is already evident in China.

The energy and emissions associated with primary steel production vary across the different process technologies but global GHG emissions associated with the steel sector were estimated as 2.6 $GtCO_2e$ in 2006 (IPCC 2014), about 25% of global industrial emissions. Steel used in buildings and infrastructure accounts for more than 50% of total production; about 25% of this is used for structural sections and around 50% for bars to reinforce concrete (Allwood & Cullen 2012). Recycling scrap steel requires 40-50% of the energy for primary production while forming and fabrication require 10-20% (Allwood *et al*. 2010)[4]. Recovered reinforcing bars need to be recycled but, given their large quantities, this still can lead to significant mitigation of global GHG emissions. Structural sections can sometimes be reformed or, better yet, re-used to achieve proportionately more reduction. However, re-use is frequently constrained by uncertainty over the original specification of the structural member (Geyer *et al*. 2002); better record-keeping is therefore needed.

GHG emissions from cement production amount to approximately 20% of global industrial emissions (Allwood & Cullen 2012), around 2 $GtCO_2e$ in 2006. They arise from energy use and also from the cement-forming calcination reactions. Almost all the cement produced is used as concrete in buildings and infrastructure. At the end of their life, cement or concrete components can rarely be re-used. The material cannot realistically be recycled but can be "downcycled" into lower-value uses, mainly as aggregate. Thus, following section 2.1, the only significant ways to reduce energy use and mitigate emissions associated with cement and concrete are to use less material in the first place (i.e. lightweighting to reduce material stock requirements; Allwood *et al*. 2011) and to extend service life (Clift and Allwood 2011).

[4] i.e. in the terms used in the Appendix, e_2/e_3 is in the range 0.4 to 0.5 whilst e_1/e_3 is in the range 0.1 to 0.2.

2.3 Material stocks and urban mining

The concept of analysing an urban metabolism in terms of flows of energy and materials was introduced in section 1.1. This is a form of material flow analysis (MFA) or substance flow analysis (SFA); the difference between the two is esoteric and will not be explored here. In the context of physical capital, the net addition to stock is the input to the urban area minus the output minus consumption; each of these components can be subdivided further, and needs to be subdivided for data compilation and interpretation (Ferrao & Fernandez 2013). One application of MFA lies in estimating the total quantity of a material, for example a valuable metal, contained in the capital stock of a city. "Urban mining" refers to the idea of thinking of this stock as a potential resource to be extracted and re-used (Graedel 2011; Brunner 2011). MFA is also used to estimate the rate at which the material will emerge into the waste stream.

Figure 2-1 illustrates how material flow accounting is applied at this level of detail. It is essential to distinguish between different ways in which the metal or material of interest is used, because different stocks have different service lives (Davis *et al*. 2007). Estimates for the quantity in each stock are obtained by integrating or summing past data for inputs over a time period corresponding to the average service life of the particular type of stock. Estimates for the quantities emerging as waste are obtained by considering the input to each stock at a past time corresponding to the mean service life.

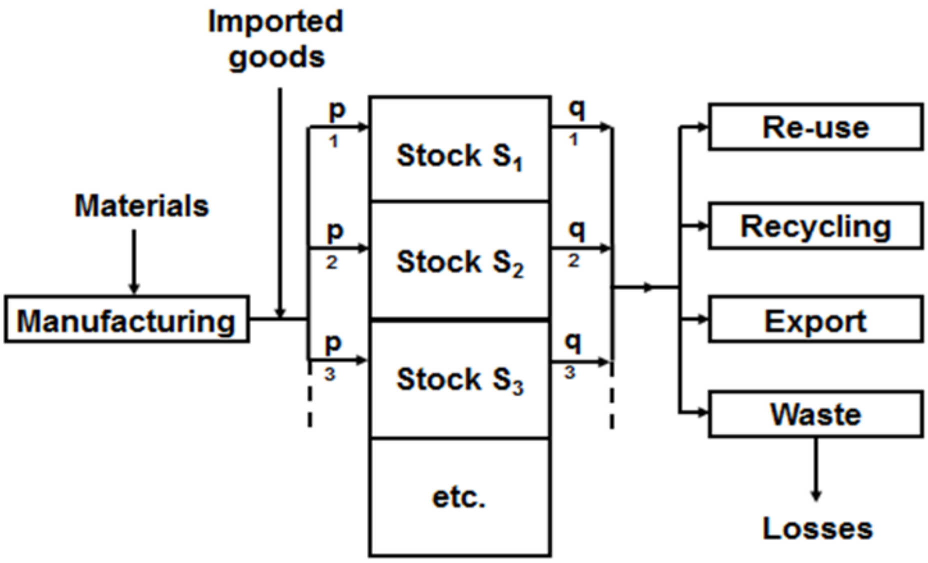

Figure 2-1. Material Flow Accounting

2.4 Impacts of urban density on urban transportation energy use and GHG emissions

Looking broadly at a set of global cities, there is an intuitive relationship between the population density of cities and the direct energy used for transportation. Sprawled, low density cites with high automobile ownership rates, encouraged by automobile dominated infrastructure systems, have high levels of per capita transportation energy use. This relationship was established in the 1990s by Australian researchers Newman & Kenworthy (1991). It has provoked a great deal of academic debate (e.g., see review by Rickwood *et al*., 2008), much of it quite futile. The simple argument is that people who are spread out in a city, living further apart, must travel further distances to interact in their daily urban activities and hence use more energy, if they can afford to. This is shown in terms of direct transportation GHG emissions per capita in Figure 2-2. Excluding New York City, the low-density North American cities have direct transportation emissions greater than 4 tCO_2e/cap whereas the higher density cities, with urbanised densities greater than 6,000 persons/km^2, have direct transportation emissions under 2 tCO_2e/cap, with the exception of Bangkok (2.3 tCO_2e/cap). There is one low density city – Amman – with transportation emissions of the order 1 tCO_2e/cap, but it is in a middle-income country.

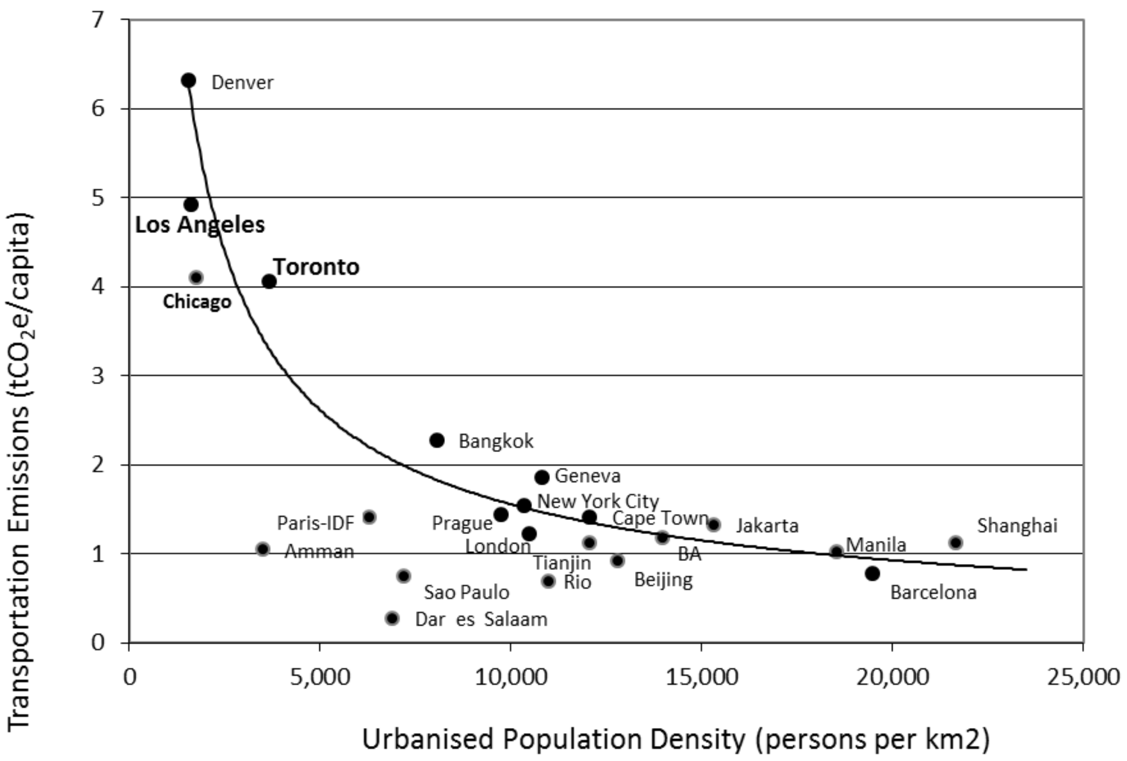

Figure 2-2. Direct per capita greenhouse gas emissions from ground transportation in relation to urbanised population density for a subset of global cities (adapted from Kennedy *et al*., 2009 using data from Kennedy *et al*., 2014)

It is important to note that urbanised density is only a significant factor in urban metabolism when looking broadly across global cities. Within either of the groupings of high or low density cities, density itself may become a secondary factor to land-use, urban design, and provision of transit in explaining emissions. This is possibly the case in UK cities, which are generally of relatively high density compared to North American cities; hence the influence of spatial structure on transportation is modest (Mitchell *et al*., 2011; Echenique *et al*., 2012). Impacts of urban density on transportation energy use in UK cities may perhaps only be significant in low density cities such as Sheffield and Middlesbrough[5].

[5] www.lincolninst.edu/subcenters/atlas-urban-expansion/google-earth-data.aspx

3. Material and energy flows: consumption and management of waste

Other factors beyond infrastructure and the built environment also influence the material and energy consumption of cities. For example, direct energy use per capita in global cities has a moderately strong relationship with urban gross domestic product. (The link between energy and GDP is sometimes used to estimate urban energy use in the absence of better data.) Another example, shown in Figure 3-1, is that per capita direct energy use for heating and industrial fuels (excluding electricity) increases with heating-degree days. In other words, colder cities use more energy for heating buildings. The scatter around the upward trend in Figure 3-1 is perhaps mostly explained by industrial combustion, with an industrial city such as Shanghai lying above the trend line and a service dominated city like Paris lying below.

While existing buildings in colder climates typically have higher energy intensity (kWh/m^2/yr), high performance green buildings can achieve low levels of energy use by employing techniques that moderate the interface between the interior and the exterior climate more efficiently. This is achieved using methods such as passive and active solar building design, high levels of insulation or nested envelopes and other techniques. Hence there is some evidence that decoupling of buildings' direct energy use and climate is being achieved for state-of-the-art green buildings (Mohareb *et al*., 2011).

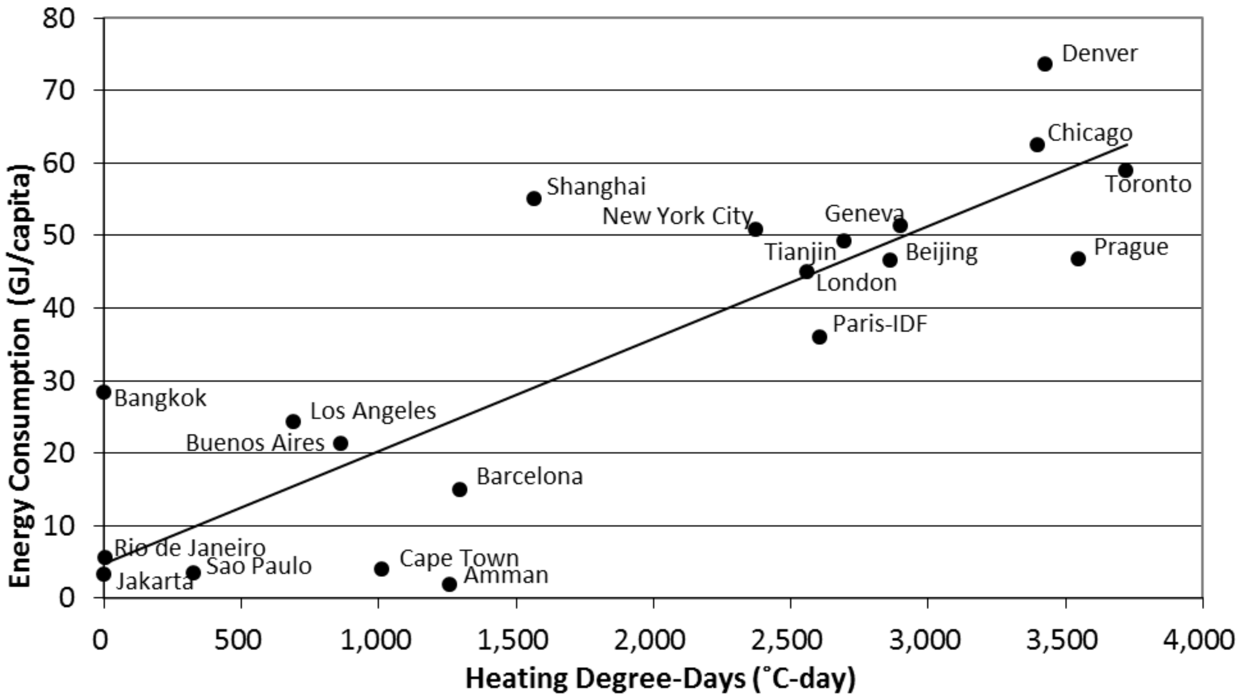

Figure 3-1. Per capita energy consumption of heating and industrial fuels in relation to heating degree-days (18-degree base) for a subset of global cities (Figure 3 in Kennedy *et al.*, 2014). Excludes electricity consumption.

In this section we first consider material and energy flows due to consumption, initially by explaining the estimation methods used and then by discussing examples of consumption accounts for UK cities.

3.1 Measurement: Environmentally-extended input-output analysis

3.1.1 Environmentally-extended input-output analysis methodology

As noted in Section 1.3.1, urban metabolism accounting according to the consumption perspective is commonly carried out using EEIO Analysis. Input-output methods have their foundation in the economics of global trade and specifically the work of Nobel Memorial prize-winning economist Wassily Leontief (Miller & Blair 2009; Leontief 1986). Leontief studied trade flows between the US and other nations, and introduced the use of input-output tables to capture how the products of one economic sector are used in the production processes of another. For example, an economy might produce a number of tonnes of steel for use in various sub-sectors such as automobile manufacturing or construction. Input-output tables therefore facilitate a quantitative analysis of how demand for a given end-use sector impacts other production sectors. They can be measured either in terms of monetary flows (MIOTs) or physical mass-based flows (PIOTs) (Weisz & Duchin 2006). Related work on the economics of urban carbon mitigation has not been reviewed here, as these studies focus on the business case for specific carbon mitigation options rather than on the physical carbon flows (Gouldson *et al*., 2014).

EEIO analysis extends economic analysis to assess the environmental impacts associated with these trade flows such as primary resources consumption (water, land, materials), energy, or emissions (Tukker *et al*., 2009; Peters *et al*., 2011). Such studies typically look at national economies and trade (CCC 2013), or household consumption (Druckman & Jackson 2009; Kerkhof *et al*. 2009). EEIO is also used to investigate the impacts of industrial sectors and companies (Murray & Wood 2010) and can analyse emissions for different types of areas according to socio-economic characteristics (Druckman & Jackson 2009). Figure 3-2 shows an example where EEIO analysis has been used to determine the total CO_2 emissions of households in London at a fine spatial scale (Super Output Areas). The authors note that the main challenge of producing such fine-grained analyses is combining “information on global production activities with information on local consumption activities”. In particular, there is often insufficient data on household expenditure data for all zones but this can be partly overcome through the use of geo-demographic clustering. This involves clustering households into a fixed number of representative categories with known average consumption patterns and then estimating the mix of these categories within each spatial zone (Minx *et al*. 2009; Druckman & Jackson 2008).

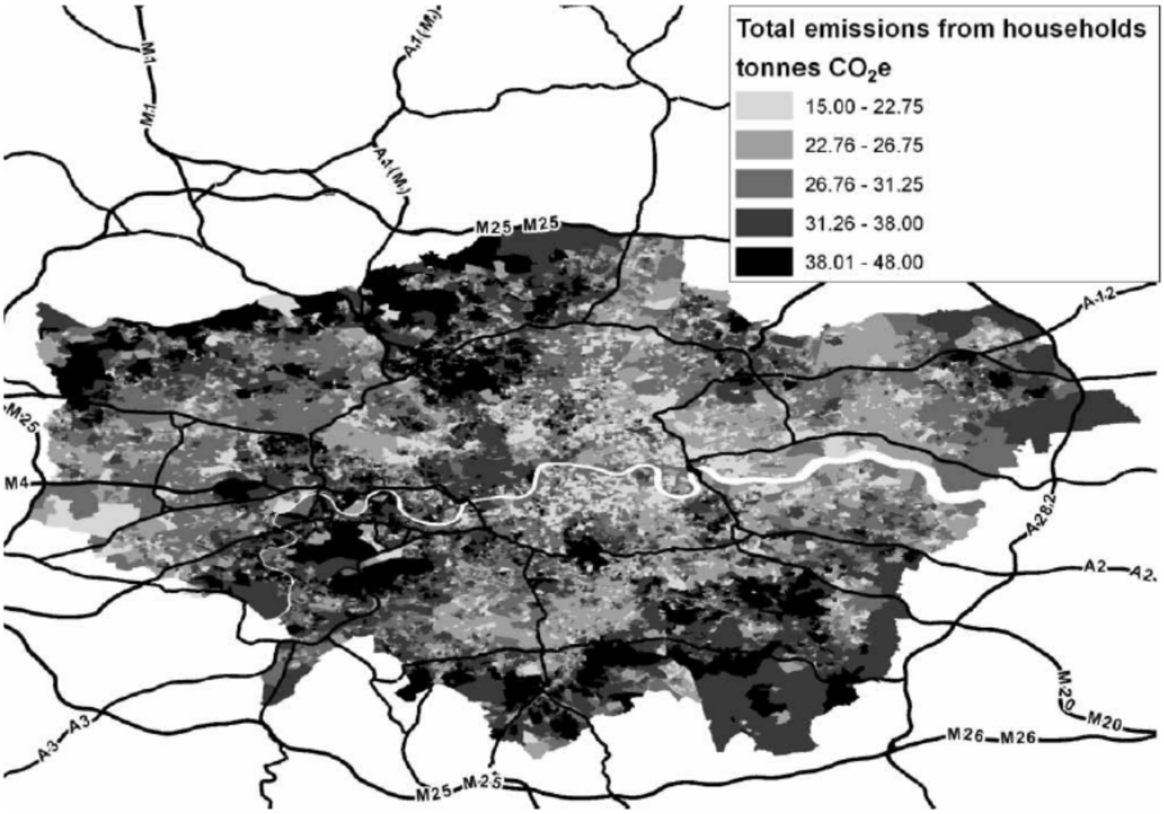

Figure 3-2. Per-household carbon footprint of London by Super Output Area in 2004 (tCO2) (Minx *et al*. 2009)

3.1.2 Data requirements

EEIO is an extremely data-intensive methodology and two main sets of data are required: (a) expenditure data by final consumers (known as 'final demand'); (b) data from which the environmental impacts of this expenditure can be estimated.

Final demand includes the expenditure of the households and government organisations within a city's boundaries. Other elements of final demand are exported goods and services[6], and investment. Final demand is considered to 'pull though' material and energy used in producing the goods and services purchased. Expenditure datasets are available from the UK Supply and Use Tables[7] at the national level; regional level expenditure for households is provided in the UK Living Costs and Food Survey and the Office for National Statistics (ONS) Family Spending reports (although these regional datasets require adjusting for use in an EEIO model[8]); and information on businesses and their expenditures at regional/local level is available from various datasets produced by the ONS such as the Business Structure Database and the Annual Respondents Database. Various small area estimation techniques exist for scaling datasets down to city level (Bradley *et al*. 2013; Chambers & Chandra 2006; Druckman & Jackson 2009; Heady *et al*. 2003; Rao 2003).

[6] As explained earlier exported goods and services are excluded from consumption account.
[7] www.statistics.gov.uk/hub/economy/national-accounts/supply-and-use-tables
[8] This is to adjust for different price bases.

Final demand expenditure is required to drive an EEIO model to estimate the material flows, energy use and emissions to be attributed to the expenditure. A convenient way to do this without developing a bespoke EEIO model is to use the Input-Output Multipliers[9]; these are published for recent years by the UK government in their guidance for GHG reporting by companies[10]. Multipliers are simple to use, but their derivation is highly data intensive. The precise data requirements of a model depend on the type of EEIO model chosen. Models can be classified as follows:

- Single region models – the economy of the system under investigation (e.g. a city) is considered to be a closed economy, i.e. to have no imports or exports. This is not suitable for application to Western cities where there are generally large material/energy flows across a city's borders.

- Two region models – the two regions are typically the city and 'the rest of the world'. Often, to reduce data requirements, the 'domestic technology assumption' is used. This assumes that imports into a city are produced with similar production and energy-conversion technologies as those produced within the city (Druckman *et al*. 2008; Tukker *et al*. 2013a; Weber & Perrels 2000).

- Multi-region models (Andrew *et al*. 2008; Lenzen *et al*. 2004a; Peters 2007; Peters & Hertwich 2006) – for a UK city, the regions would typically be the city under study, the 'rest of the UK', and then the UK's main trading partners, which may be represented by any number of regions from three to 100 or more.

In each region, industry is represented by 'sectors' (such as the agriculture sector, mining sector and electrical power generation) and information is required detailing the material/energy use and/or emissions from each industry sector; how the sectors within that region trade with each other, and also how they trade with sectors in other regions. This gives an indication of the 'recipe' used to produce goods and services with that country. Such data is generally provided by national governments in Input-Output Tables, and datasets are only very rarely available for small regions or at city level (Beyers 2013; Lenzen *et al*., 2004b). The UK tables have around 120 sectors, US tables have over 350 sectors and Japan tables have over 400 sectors. A fundamental assumption in EEIO is that sectors are assumed to be homogenous. For example the UK national Input-Output Tables feature a single electrical power generating sector; this means that the same technology is assumed to be used for producing power from nuclear, wave, solar and gas. This clearly gives rise to inaccuracies, and generally the fewer sectors provided, the larger the inaccuracies.

Input-Output Tables are costly and time-consuming to produce. Under EU regulations member states are only required to produce Input-Output Tables at 5 year intervals (European Commission 2007). Therefore analysis is often carried out using out-of-date tables. In recent years a number of multi-regional datasets have been developed that bring together Input-Output Tables produced by multiple countries (which are often for different years and for different sector aggregations). These include Eora (Lenzen *et al*. 2012); 'The World Input-Output

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[9] www.scotland.gov.uk/Topics/Statistics/Browse/Economy/input-output

[10] See e.g. Environmental Reporting Guidelines at www.gov.uk/government/uploads/system/uploads/attachment_data/file/206392/pb13944-env-reporting-guidance.pdf

Database (Timmer 2012); and EXIOPOL (Tukker *et al*. 2013b). These datasets make multi-regional modelling less laborious than previously.

It is important to remember that EEIO modelling only estimates the supply chain material flows/energy use GHG emissions due to final demand expenditure. It does not capture direct material flows, energy use or associated GHG emissions. For the UK these are published in the Environmental Accounts (ONS 2013), with some data being available at local level.[11] Data are provided for European countries by Eurostat (Eurostat 2014). These direct flows must be added separately to estimate total consumption perspective results.

3.2 Material and energy flows: examples of consumption accounts for UK cities

Recent years have seen numerous studies of urban environmental impacts, be it comparative analyses of urban metabolism in general (Kennedy *et al*. 2007) or more detailed studies of single resource or pollutant flows (e.g. Brown *et al*. 2008). This section provides a brief review of the material and energy flow studies that have been conducted for UK cities.

Notwithstanding the fundamental problems with the Ecological Footprint (EF) as a metric (see Section 1.3.3, pp.20-21) many UK studies have used the EF methodology. One study estimates that, between 1999 and 2004, 60 to 70 studies of UK EFs were conducted, having strong interest from local authorities as well as regional and devolved administrations (Collins & Flynn 2007). These studies were motivated by two drivers: the difficulty of communicating the connections between global sustainability aspirations and local policy, and the availability of research funding for resource flow studies following the 1996 introduction of a landfill tax credit scheme. Although the EF technique can be helpful for public engagement and communication, its use as a robust metric for scientific impact assessment and accounting is questionable, as discussed in Section 1.3.2 (Opschoor 2000; Ayres 2000; Rees 2000). In some of these studies, the EF is supported by a detailed urban metabolism analysis (e.g. London); however in other cases, only the headline EF measure is presented and these studies are of limited use for understanding urban metabolisms.

3.2.1 London

One of the most detailed UK studies of urban resource consumption is the Best Foot Forward (BFF) analysis of London (BFF 2002a). The project was funded as part of Biffa Waste Management's Biffaward scheme, with the support of the Chartered Institution of Wastes Management, the Greater London Authority, and the Institution of Civil Engineers. Introducing the report, then Mayor of London Ken Livingstone noted that the study marked "the first such [footprint] analysis of a major world city" and highlighted how the resulting information on London's metabolism would be used to guide spatial planning policy as well as specific environmental strategies on air quality, biodiversity, energy, noise, and waste management.

[11] See for example: www.gov.uk/government/statistical-data-sets/sub-national-energy-consumption-statistics; http://data.gov.uk/dataset/electricity_consumption_at_local_authority_level;

The overall goals of the study were:

- "To quantify and catalogue the energy and materials consumed by London and Londoners, and where possible map the flows of these resources.
- To calculate the ecological footprint of the citizens of London.
- To compare the ecological footprint of Londoners with other regions.
- To compare the ecological footprint of Londoners with the globally available 'earth share' to estimate ecological sustainability.
- To quantify the ecological sustainability of a range of improvement scenarios.
- To assess the availability and quality of data required to carry out this type of analysis, and in certain instances make recommendations to improve data requirements for resource flow and ecological footprint analyses"

To achieve these aims, the researchers gathered data on annual resource flows for the year 2000 in the following categories: direct energy consumption, materials, waste, water, and land use. Additional contextual data relating to tourism, transport, and demographics were also collected. This information came from databases curated by BFF and over 240 individuals and organisations identified during project consultations. Data quality and availability were judged to be very good overall although coverage was not equal in all sectors and in some cases, national data had to be downscaled to local level, e.g. on a per capita or gross domestic product basis.

Once these urban metabolism accounts were created, the data were converted into a headline EF metric measured in global hectares (1 gha = 1 hectare of land with global average productivity and waste assimilation capacity) (Rees 1996; Moran *et al*. 2008). Complementary footprints were also calculated for specific end-use sectors, for example, the footprint associated with car travel measured in gha per passenger-km. Finally the footprints associated with three alternative consumption scenarios were calculated: 'business-as-usual' (current trends extended to 2020), 'evolutionary' (impact of current policy targets), 'revolutionary' (2020 consumption patterns consistent with "one-planet" living in 2050). A similar set of scenarios for London are described in more detail in Girardet (2006).

The headline results were as follows and as shown in Figure 3-3:

- Annual per capita resource consumption of: 20.3 MWh of energy, 5.5 tonnes CO_2 emitted, 6.7 tonnes of material consumed of which 57% is construction material, 118 thousand litres of water consumed (28% of which is leakage)
- 26 million tonnes of waste generated, 58% from the construction and demolition sectors, 13% from the domestic sector
- An EF of 6.6 gha per capita, slightly higher than the 6.3 gha per capita UK average, and much greater than the 2.18 gha per capita estimated as a global average availability.
- For long-term sustainability, it was estimated that a 35% reduction in EF will be needed by 2020, and an 80% reduction by 2050.

- While more research and better data would improve the accounts, the report's authors felt there was "already enough data in the public domain to indicate reliably that London lifestyles are not currently sustainable."

The report was also followed by a more detailed examination of the actions that the public and private sector could take to reduce London's EF, targeting interventions in resource use (e.g. increased recycled content in short-use goods), food, energy and the built environment, and personal mobility (London Remade 2004). Additionally the GLA commissioned a review of the report that highlighted three shortcomings: poor comparability over time, limited accounting for the activities of commuters into London, and only very broad consideration of the service sector, a major contribution to London's economy (GLA Economics 2003).

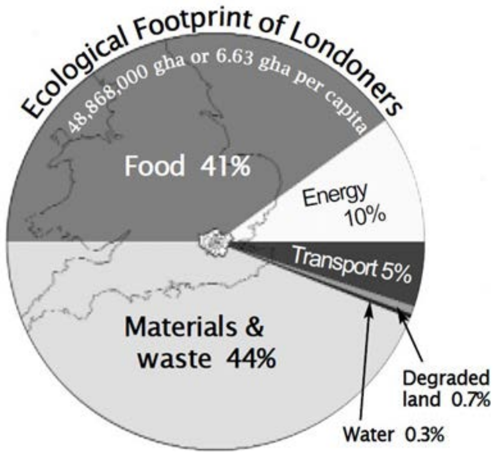

Figure 3-3. Ecological footprint of Londoners, by component, showing actual size and the UK (BFF 2002).

3.2.2 Other cities

The original Biffaward scheme (see Section 3.2.1) supported 554 projects on resource efficiency around the UK; many of these were ecological footprint studies, without detailed supporting data on urban metabolic flows (London, York, and to a lesser extent Angus being the exceptions). However to highlight the breadth of work that has occurred to date, we consider these EF studies here, using the gha indicator for comparison purposes.

By 2007, as part of the scheme, local authorities had completed or commissioned studies for the Isle of Wight, London, York, Cardiff, North and North East Lincolnshire, Greater Nottinghamshire, Aberdeen City, Aberdeenshire and North Lanarkshire (Collins & Flynn 2007). Browne *et al*. (2012) point to these studies and other examples including:

- A material flow analysis and EF of York (Barrett *et al*. 2002) found that the total material requirement for York was 18.8 tonnes per resident, of which 46% was actual material and the rest was energy carriers or 'hidden' flows that do not enter the economy (e.g. waste trimming from forestry). About two-thirds of these material requirements were attributed to the construction of roads and houses. The total EF was found to be 6.98 gha per capita, greater than the UK average of 6.3. A variety of policy scenarios are proposed as well to try and achieve the goal of 'one planet living'; these relied upon greater self-sufficiency and increased efficiency of resource use (i.e. reduced material and energy consumption).

- The study of the EF of transport in Merseyside (Barrett & Scott 2003) showed that existing policy initiatives were likely to lead to an increased environmental impact, but the footprint measure could be used in public education programmes to promote alternative modes of travel.

- A study of Bath's EF did not present the supporting material flows in detail, but does contain an extended discussion of how the headline ecological indicator should be interpreted in local context (Doughty & Hammond 2004). They focus on the history of the city of Bath, its unique character, and the benefits of having a unitary authority that can coordinate relevant policies in both the city and its hinterland.

- The Scottish Executive commissioned an EF analysis for five cities: Aberdeen, Edinburgh, Dundee, Inverness, and Glasgow (BFF 2002b). The results showed all these cities with a greater footprint than Scotland's average biocapacity of 4.52 gha per capita (BFF 2004). Angus Council also commissioned a footprinting study and found a footprint of 4.78 gha per capita (BFF 2003).

- In Wales, an EF analysis was conducted for Cardiff Council (Collins *et al*. 2005; Cardiff Council 2005). The results showed a total EF of 5.59 gha per capita, mainly associated with food and drink (24%), energy use (18%), personal transport (18%), and infrastructure and housing (16%). The study also included a detailed case study of the impact of the 2004 FA Cup Final, highlighting the importance of personal transport, food, and packaging waste.

Figure 3-4 compares the ecological footprints of different UK cities and regions.[12]

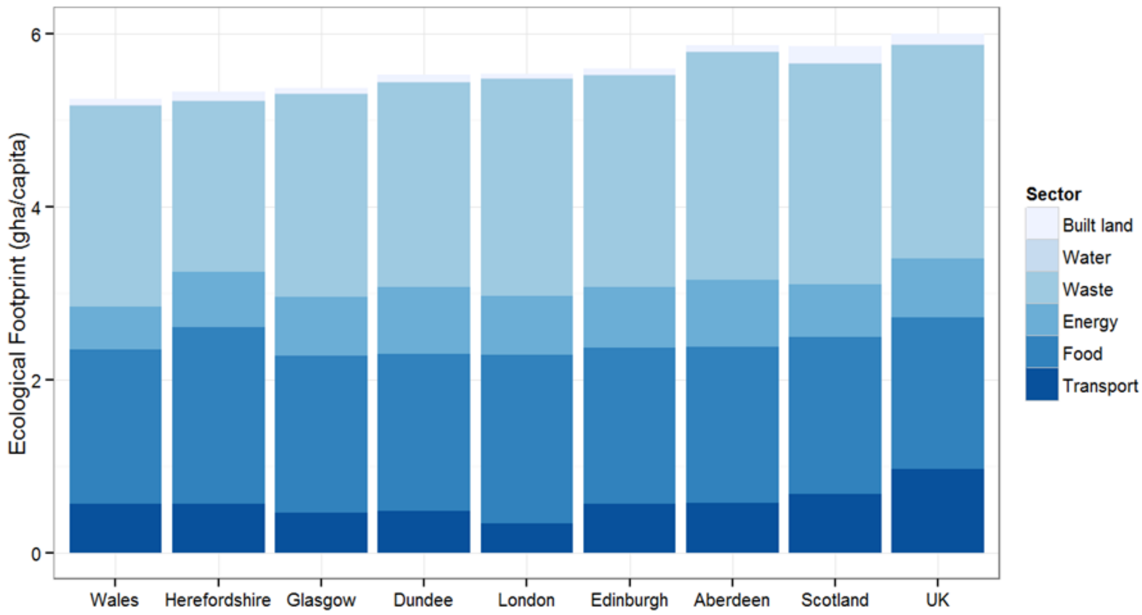

Figure 3-4. Ecological footprints of selected UK cities/regions (data from The Scottish government 2003)

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[12] Some of the plotted values do not match those reported elsewhere in this report due to varying assumptions about what contributes to the total footprint; this reinforces the difficulty of using ecological footprints for comparative analysis.

In addition to these city-specific studies, other research has provided spatially-disaggregated measures of resource consumption that could inform urban metabolism studies for any local area in the UK. The first example is the spatially-disaggregated carbon footprint analysis presented in Figure 3-2. The same input-output techniques were also used in a study of water footprints that calculated both internal and external water consumption (i.e. water consumed from UK resources versus global supplies) (Figure 3-5). Methodologically, these approaches are more efficient as they build from official statistics with national coverage, irrespective of whether a spatial region is classified as urban or rural. One such recent report highlighted that, using these methods, 90% of human settlements in the UK can be classified as net importers of CO_2 emissions. The study also demonstrates that the links between demographic, economic, spatial characteristics of settlements and CO_2 emissions are complex (Minx *et al*., 2013).

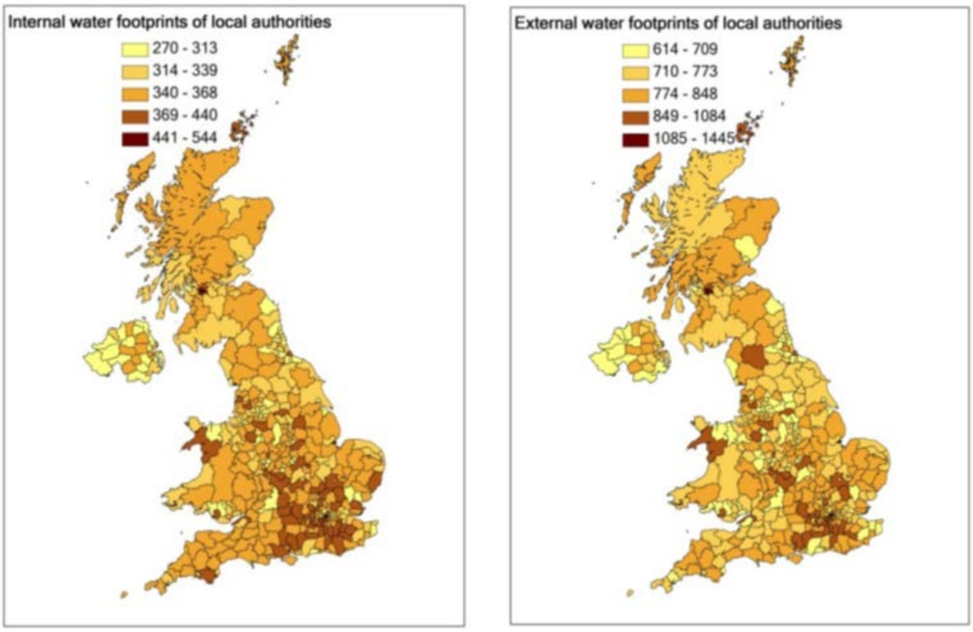

Figure 3-5. Spatially-disaggregated water footprints for UK local authorities (m3 per person-year) (Feng *et al*. 2011)

3.3 Waste and management

It is common to distinguish between different forms of solid waste arising from urban areas:

- construction and demolition waste (C&DW)
- commercial and industrial waste (C&IW)
- domestic – municipal solid waste (MSW)

Data on waste arisings for the UK are more complete than for some countries in the EU, but less complete and current than in other member states and, for detailed planning purposes, need to be more complete and current.

In addition to these forms of waste, it is necessary to manage end-of-life products, particularly electrical and electronic devices (including domestic and office appliances) and vehicles, which are the topics of specific legislation in the EU (See Section 2.3). Urban structures must include systems to collect and manage the waste.

MSW and C&IW represent materials with very short service lives; therefore the waste arisings are effectively an immediate response to consumption and can be estimated from current consumption (see Section 2.2). The flow of waste for final disposal can be reduced by reducing consumption, or by re-using or recycling materials from the waste. Re-use and recycling are particularly appropriate for packaging waste. Deposit-and-return-systems to encourage recovery of packaging are therefore an important part of the urban infrastructure. The Swedish "Returnpack" system, under which bottles and cans are returned via retail outlets, has proved to be particularly effective; it has the capacity to return containers to their suppliers so that it can enable re-use (i.e. refilling of bottles), although re-use has been phased out in recent years in favour of recycling. This is a further illustration of the importance of re-use and refurbishment of durable goods, explained in Sections 2.2 and 2.3 and in the Appendix. Effective recycling requires different materials to be kept separated at source or commingled waste to be separated on receipt at a transfer station (POST-2013). Whether source separation or mechanical separation is more effective is an open question but, either way, transport of separated waste or facilities to separate waste materials are essential parts of the urban infrastructure.

The notion that cities might develop with circular metabolism was popularised by Giradet (1992), although there are limits to the extent to which notions of the circular economy can in practice be applied (Ayres, 1997; Allenby, 1999; Allwood, 2014). In comparison to rural areas, the high volumes and close proximity of activities in cities does provide greater economic opportunity for re-use or recycling of materials. Desroches (2002) provides an insightful historical review of such industrial symbiosis, examining four urban regions where significant inter-industry waste exchange is practiced. He, moreover, cites several authors from the 19th and early 20th century whose writings suggest that urban industrial symbiosis used to be relatively common. Desroches thus argues that industrial symbiosis is a type of agglomeration effect, and "The fact that cities or regional economies… have probably always exhibited localised inter-industry recycling linkages seems highly plausible." A recent application of large-scale urban symbiosis is the example of Kanagawa, Japan (Van Berkel *et al*., 2009); and the potential to design city neighbourhoods with a greater degree of circularity of materials has been discussed by Kennedy *et al*. (2011). Nonetheless, there are some materials within urban metabolism for which recycling is inherently infeasible or would require so much energy as to be undesirable (Allwood, 2014). The limits to notions of the circular economy were worked out by pioneering industrial ecologists such as Ayres (1997) and Allenby (1999), both of whom provide categories of materials that can or cannot be recycled.

However much material is recovered for recycling from MSW and C&IW, there is inevitably a proportion which cannot be recycled (Allwood 2014) due to contamination or degradation during reprocessing. For paper, as a specific example, recycling damages the fibres so that recycling systems need to ensure a sufficient input of non-recycled paper to maintain the paper properties (Hart *et al*. 2005). While new technologies may reduce damage and contamination and thereby increase the proportion of material recycled with environmental and economic advantage, it is an inevitable thermodynamic conclusion that total recycling is an impossibility (Allwood 2014). The material that must be "purged" from recycle loops includes mixed or contaminated plastics, paper and card and also the organic fraction, mainly food waste. Organic waste can be composted to reduce its volume, but demand for compost is limited. Energy recovery therefore represents the most beneficial use of the residual waste. The relative benefits of energy and material recovery should be assessed by life cycle assessment (see Section 1.3.3). A number of software packages are available for this kind of assessment; one of the most widely used is WRATE, originally developed by the Environment Agency but now maintained by AEA Technology (2012). Comparison of energy-from-waste technologies on a life cycle basis shows that the efficiency of energy recovery is the most significant consideration, with material recovery of secondary concern. Some technologies currently under development also enable higher levels of metal recovery, primarily because materials are recovered in the pre-treatment stage before thermal processing (Evangelisti *et al*. 2014c); however the quantities available for recovery in this way are nugatory in the context of overall material flows. Energy may be recovered by some form of thermal processing (i.e. combustion, gasification or pyrolysis) in a plant of sufficient scale to be equipped with gas cleaning equipment adequate to bring the flue gases to acceptable standards. The organic fraction can also be treated by anaerobic digestion (AD) to produce a methane-rich biogas. The raw biogas can be used close to where it is generated, or can be upgraded to biomethane with properties similar to natural gas and distributed along with natural gas.

Elsewhere in Northern Europe, energy is routinely recovered from solid waste in combined heat and power (CHP) plants, with the heat distributed, primarily for space and water-heating. Where there is no demand for the heat, usually because heat distribution to domestic and commercial buildings is not established practice, energy recovery from waste is limited to electrical power. Producing only electricity reduces the total energy recovery by a factor of three or more. This provides a specific example of the importance of providing for heat (and cooling) distribution in the urban metabolism.

Recovering energy from waste, whether by thermal treatment or AD, avoids the emissions of methane (with its much higher greenhouse warming potential compared to carbon dioxide) which arise when degradable waste is landfilled and also offsets GHG emissions from use of fossil fuels to provide electricity and/or heat. Furthermore, organic waste is classified as “renewable” because its carbon content is mainly part of the renewable carbon cycle, so that the carbon dioxide emissions from using organic waste as an energy source do not contribute to the GHG balance. These elementary considerations underline the importance of treating waste flows as integral parts of the urban metabolism. The contribution of energy-from-waste to GHG mitigation can be significant, even though the proportion of energy consumption which can be met from waste is not large. For example, Evangelisti *et al*. (2014a) considered, as a specific case study, energy recovery from the organic fraction of municipal waste in the London Borough of Greenwich. AD leads to more GHG savings than either incineration or landfilling with energy recovery from the landfill gas (Evangelisti *et al*., 2014a). If the biogas is upgraded, distributed via the existing gas grid and used in micro-CHP units at the scale of single dwellings, it can provide up to 4% of the total energy demand of the Borough and save up to 12 tCO_2e pa per dwelling equipped with micro-CHP (Evangelisti *et al*., 2014b).

The other forms of waste – construction and demolition waste (C&DW) and end of life manufactured products – can most effectively be reduced by extending the service lives of buildings and products (see Section 2.3). Given their long service lives, estimating arisings of waste appliances and vehicles requires data in the form of time series for historical consumption (see Section 2.2). Re-use and recycling are important in managing end-of-life products to minimise demand for new materials (see Section 2.3), with specific measures needed for particular products and materials where the materials have sufficiently high value, but these considerations are not exclusive to the metabolism of cities. Arisings of construction and demolition waste depend mainly on the economic cycle. Apart from specific components and materials which may have sufficient value for recovery to be economically attractive, demolition and construction waste is of much lower value and can only be recycled in low-value uses, primarily aggregate and hard-core used in infrastructure construction. The timing of infrastructure projects will not always coincide with the arisings of demolition and construction waste. Therefore it is necessary to provide buffer storage capacity to ensure full use of D&CW.

As a specific illustration, lack of such capacity has been one of the main barriers to re-use of road surfacing materials (Elghali 2002).

3.4 Behavioural change

‘Behavioural change’ in the context of urban metabolism refers to the design and implementation of policies to influence the consumption practices and choices of citizens. All policies aim at some form of behavioural change, of course (Dolan *et al*., 2010; House of Lords, 2011); but there is a particular aspect to the goal of behaviour change for sustainable development. In policy debates over sustainable development, the focus on behavioural change relates to the perceived need for action to modify consumption patterns, perhaps very radically, as well as for policies to change production systems, for example in order to reduce greenhouse gas emissions and improve energy efficiency in households (Jackson 2005, 2009; Corner 2012).

The core idea here is that relying on technological innovation and supply-side measures to achieve sustainable development cannot be enough. Changes are required on the demand side, for a range of reasons:

- first, it is highly unlikely that supply side innovations can bring about all the reductions in unsustainable resource use and emissions that we need (Jackson, 2009);
- second, widespread adoption of new technologies requires in many cases changes in values, attitudes and practices by citizens ; and
- third, technological transformations in democratic societies should be based on consent and engagement with citizens – for example, ‘smart city‘ innovations are increasingly seen to be dependent on meaningful and effective engagement with and feedback from citizens (Smedley, 2013).

Moreover, policy measures from the demand side (regulation and Green procurement) are crucial drivers of innovation for the Green economy (Hopwood *et al*., 2005).

There has been a rapid growth of interest among policy-makers and researchers in the application of social psychological insights to policy design and development. There is a recognition of the inadequacy of ‘technological fixes’ in promoting sustainable development and an awareness that many policy problems require changes in behaviour, attitudes and values on the part of citizens, if they are to be tackled effectively (Dolan *et al*. 2010; House of Lords 2011; Maio *et al*. 2007). This raises the complex and unsolved problems of how behaviour, values and attitudes are related, and how policy-makers can proceed in promoting behavioural change in general, and in relation to ‘sustainable living’ in particular.

Cities are the classical locus for the formation of citizens and solutions to collective action problems (Jacobs 1961; Carley *et al*. 2001; Bell & De-Shalit 2011; Barber 2013). Moreover, they are the site of most dynamic economic, social and environmental developments and pressures in the world, and can be seen as emergent ecosystems in their own right (Elmqvist *et al*. 2013). As such, cities have the potential to be the dominant setting in which social innovations for sustainable development will emerge, although there are major barriers confronting cities in realising this potential (see Bai, 2007, on arguments concerning scale and readiness in urban environmental management).

Below we outline the main strands of current thinking on behavioural change for sustainability, and we discuss the challenges of promoting and monitoring changes in behaviour. Examples are drawn from current and recent research programmes on behaviour, values and lifestyle change. We conclude by highlighting the importance of cities as enabling environments for behavioural change (Levett & Christie 1999; Birkeland 2009).

3.4.1 Theories of behaviour change

Theories of behaviour change are diverse and contested, but some widely accepted conclusions can be identified. In this section we outline key points arising from recent research and policy debate (see e.g. Dolan *et al*. 2010; House of Lords 2011; Corner 2012; Jackson 2005, Maio *et al*. 2007). We also highlight two general approaches to behavioural change which have parallels with broad conceptualisations of the city as a system, and which are often seen as opposed, but which need to be brought together for an holistic view of how people live and how they can be influenced to live differently in pursuit of sustainable development.

No generally agreed theory of human behaviour exists on which to found a policy-oriented theory of behavioural change. Many theories are available, but all are contested. Major contributions include the theory of planned behaviour (Ajzen 1991); the theory of structuration (Giddens 1984); theories of social practice (Shove 2008, 2010; Shove *et al*. 2012); theories based on the activation of norms and values (Jackson 2005). So we have no universal theory of why we consume in the ways we do, what constrains us and what might promote more 'pro-environmental' living and sustainable consumption practices. However, we have a reasonable level of consensus on key features of behaviour in relation to consumption and the factors involved in promoting and constraining changes in everyday practices (Jackson 2005; Corner 2012; Spence & Pidgeon 2009; Upham *et al*. 2009). These aspects include the following:

- 'rational choice' models of human behaviour are inadequate for explaining the way in which we make decisions: these rely on a model of the individual which does not take sufficient account of the social dimensions of choice and practices in everyday life;

- decisions and behaviours are guided and shaped to a large extent by social norms, which are embodied in the habits we acquire, the expectations we have of how others are likely to behave, and imitation of what others do and desire;

- the power of social context is crucial in understanding behaviour and the scope for behavioural change: even when a person's attitudes and values are conducive to pro-environmental behaviour, say, their realisation in everyday activity is constrained by what the prevailing norms and context allow, and by 'lock-in' of behaviour through lack of capacity to change – for example, because of lack of money, or non-availability of infrastructure (affordable and accessible public transport, for example).

In the light of this, an individualistic approach to policy design is unlikely to work: for example, simply providing information about potential changes in consumption to individuals and households, and hoping this information will be the catalyst for action, will in most cases not work. Other supportive measures are needed to provide the capabilities and contexts in which people will be willing to act:

- these measures can include financial and other incentives that appeal to self-interest (for example, saving money from energy efficiency investments). However, these incentives are often insufficient to overcome the obstacles to take-up of more sustainable behaviours;

- appeals to change behaviour based on values that are important to a citizen or a community, and that are accompanied by enabling measures, may be more effective in catalysing willingness to act (Crompton & Kasser 2009; Corner 2012);

- policies to inform, influence and change individuals' behaviour therefore need to be carefully framed to work on, or to provide, the social context in which more sustainable lifestyles can be cultivated. This implies the importance of a focus on people acting in a community of interest, so that supportive social norms and feedbacks can be generated. As Jackson (2005) notes, *'Individual behaviours are shaped and constrained by social norms and expectations. Negotiating change is best pursued at the level of groups and communities. Social support is particularly vital in breaking habits and in devising new social norms'*.

Two broad theoretical approaches to behavioural change in the context of sustainable living can be identified. One is a sociological, 'structuralist' theory of social change, in which large-scale features of infrastructure and social systems constrain and generate individual actions. The other is a social-psychological approach to change, in which individual agency, however constrained, remains significant, and in which activation of new values and attitudes is important to influencing behaviour. These two approaches can be viewed as antithetical (Shove, 2010) or as complementary (Whitmarsh *et al*. 2011). We take the latter view, and relate these two approaches below to wider conceptions of the nature of urban systems.

The 'structuralist' approach noted above is a major feature of the influential theory of social practice (Shove and Spurling, 2013; Shove *et al*. 2012; Shove 2008). In this framework, values, attitudes and individual agency are downplayed to a large extent in favour of the power of structures and systems. Consumption is seen to be the realm of practices – routines of regularised behaviours and habits – which are generated by the social systems of norms, timetables and rules all around us, and by the nature of the infrastructures with which we interact. In strong forms of this approach, individuals seem to be reduced to 'carriers of practices' (Shove *et al*. 2012) and policy based on changing values and attitudes in the hope of influencing behaviour is seen as wrong-headed. The point is to change the systemic features of everyday life – technologies, standards, laws, conventions, infrastructures – that generate practices. This approach is compelling in that it articulates powerfully the degree to which individuals are constrained in the choices they make and to which routines and habits reproduce practices. The theory of practice can also be seen to be complementary to a view of the city as a metabolic and complex system in which individuals are cells and transmitters of activities, with the system as whole and its structural features having far more 'agency' than any component.

Where the theory of social practice can be challenged is in relation to the possibility of change and the need for open deliberation and democratic consent. Using a strong account of practices, in which human beings are characterised as bearers of habits and their values, and attitudes are of little account in effecting changes in practices, it becomes hard to see how any change can be brought about by individuals. Yet clearly people do change their values from time to time and do change their practices consciously as a result, and moreover from a normative democratic perspective we should also wish that individuals should have 'some self-direction of their own behaviour' (Whitmarsh *et al*. 2011; Sayer, 2013). Even if we take the (very debatable) view that most people are acting unreflectively, most or all of the time, as carriers of practices and nodes in a vast system of structural determinants of behaviour, we note that the latter system of structural determinants is also subject to change based on shifts in values, attitudes, choices and goals by elite decision-makers, whose agency does count. So practice theory, in its strong forms at least, is as incomplete as would be a strongly individualistic theory of behaviour based on an economistic view of rational choice.

The second theoretical approach is a fusion of the individualistic and structural perspectives (see e.g. Gatersleben & Vlek 1998; Corner 2012; Jackson 2005). It takes the view that we do indeed spend much of our time following structurally constrained routines, habits and practices, and that much of our behaviour is framed by social context and available norms. But it also sees change as generated in certain conditions by events, interactions and incentives that enable an iterative process of imitation, discussion and commitment by people in groups – as in community-level action for promotion of pro-environmental behaviours and values (Peters *et al*. 2010; Smith *et al*. 2013; Seyfang 2009; White & Stirling 2013). This approach is one in which the structural and systemic vision of society – or of the city, more narrowly – is complemented by a view of the possibility and desirability of human agency for change, based on co-operative action and feedback in community. In relation to urban metabolism, this is a view that insists on the identity of the city not only as complex system (Bettencourt 2013; Bettencourt *et al*. 2007) but also as polis, a decision-making community whose diverse members can collaborate to change the conditions in which they live together (Levett & Christie 1999; Bell & de-Shalit 2011; Barber 2013).

Jackson (2005) notes that a grand unified theory of behaviour, bringing together structuralist practice-based accounts with more individual agency-based accounts, may be impossible to devise, but that a necessarily incomplete pragmatic synthesis is nonetheless available, a view echoed by Whitmarsh *et al*. (2011). Jackson (2005, p.6) sums this up:

> *"...my behaviour in any particular situation is a function partly of my attitudes and intentions, partly of my habitual responses, and partly of the situational constraints and conditions under which I operate. My intentions in their turn are influenced by social, normative and affective factors as well as by rational deliberations. I am neither fully deliberative nor fully automatic in this view. I am neither fully autonomous nor entirely social. My behaviours are influenced by my moral beliefs, but the impact of these is moderated both by my emotional drives and my cognitive limitations."*

The theoretical approach we adopt to human behaviour will influence the policies we decide to adopt in promoting behaviour change. This is the subject of the next section.

3.4.2 Promoting behaviour changes

There is a consensus among proponents of both practice theory and more social-psychological approaches to behavioural change that an individualistic model for behavioural change policy is inadequate (Jackson, 2005; Corner, 2012; Shove, 2010). Policies that rely on provision of information as a prompt for new choices by individual consumers will not '*do justice to the complexity of consumer behaviour nor exhaust the possibilities for policy intervention in pursuit of behavioural change*' (Jackson 2005, p7). A summary of potential interventions to promote a) better understanding of the dynamics of behavioural and value change, and b) possibly lasting and effective changes is given below, drawing on a range of recent syntheses (Jackson 2005; Corner 2012; Dolan *et al*. 2010; Crompton & Kasser 2009; Maio *et al*. 2007; Whitmarsh *et al*. 2011):

- to understand the dynamics of change, and the conditions in which interventions might work or not, we need more systematic experimentation. A currently popular approach in government is the design of randomised controlled trial (RCT) studies to test the effectiveness of interventions using a target sample and a control group (Dolan *et al*. 2010). This approach is valuable, but it must be noted that more qualitative approaches are also needed in order to explore motivations, narratives and constraints in households and communities;

- practice theorists and social-psychological behaviour change theorists agree on the importance of structural changes to overcome 'lock-in' of people to unsustainable behaviours: these might include investment in urban infrastructure to enable more waste recycling and re-use; more public transport use; more cycling and walking to work, school and amenities; and more take-up of renewable energy services. They would also encompass changes in regulatory systems, product and service standards, procurement systems; and they would also focus on sustainable development changes via land use planning and urban design to favour greater conviviality and mutually reinforcing good practices in city communities (Jacobs 1961; Hall & Ward 1998; Levett & Christie 1999);

- there is growing interest in the idea that people might be more open to initiatives and incentives for sustainable living at moments of significant transition in their lives, when habitual practices are liable to be disrupted and when new practices could be adopted (Verplanken & Wood, 2006). This is being explored for example in projects within the DEFRA-funded Sustainable Lifestyles Research Group programme by the University of Surrey and other academic partners (see www.sustainablelifestyles.ac.uk). Moments of transition that are being researched include moving to a new home; the arrival of the first child in a household; and entry into retirement;

Community-based projects to encourage new sustainable lifestyles or particular practices are important in the light of the discussion of behaviour change theory above. Many initiatives have explored the scope for promoting behaviour change through engagement of people in local communities and in communities of interest (Peters *et al*. 2010; Smith *et al*. 2013; Seyfang 2009; White & Stirling, 2013). Areas of interest for this kind of approach include community energy schemes, local food-growing projects, local waste recycling and re-use projects. One implication of the core insights of recent social psychological theory and experiment, concerning the powerful role of social context in individual practice and decision-making, is that people are more likely to become interested in change, and to stick with changed lifestyle practices, if engaged in a collective initiative with peers (such as neighbours). See for example the DEFRA-WWF study on Community Learning and Action for Sustainable Living (CLASL), which was based on facilitated activities by citizens related to church and school communities, both good examples of congregational settings in which mutually reinforcing changes in practices, norms and values could be established (Warburton 2008);

Wider community initiatives to engage populations on a large scale with experiments and interventions for promotion of sustainable living have great relevance for urban policy, given the scope for dense connections and feedbacks in city networks. A notable example of city-wide social experimentation with behaviour change in mind is the Binghamton Neighbourhood Project in New York State, run by Binghamton University with stakeholder organisations across sectors in the city (see http://bnp.binghamton.edu/).

No ready toolkit exists for policy-makers based on the emerging social science of behaviour change for sustainable living. However, it seems clear that a diverse range of interventions is required in order to overcome the limitations of the individualist approach to behaviour change and to tackle structural constraints as well as facilitate action by people acting in meaningful concert at the level of communities of place and interest. This means a mix of 'upstream' (structural) and ‘downstream’ (local and individual/community focus) measures (Maio *et al*., 2007). Jackson (2005, p6) sums up the need for a strategy using a range of approaches to promoting behavioural change:

> *“A concerted strategy is needed to make behaviour change easy: ensuring that incentive structures and institutional rules favour pro-environmental behaviour, enabling access to pro-environmental choice, engaging people in initiatives to help themselves, and exemplifying the desired changes within government’s own policies and practices.”*

3.4.3. Monitoring behaviour changes

Monitoring behaviour change is not straightforward. It depends on a) effective measurement of baseline conditions (consumption indicators, self-reported values and attitudes and behaviour, context), and b) supplementing self-reported changes by citizens with robust measures of actual shifts in consumption over time. Methods for isolating the effects of particular interventions, for example via RCTs (Dolan *et al*. 2010), are needed for rigorous evaluation, but are not always practicable. Moreover, much behavioural change that could be valuable for sustainable development, such as local innovations from grassroots initiatives in (for example) food-growing, might well emerge in unforeseeable ways from communities and informal projects, and be less amenable to the kind of specification of inputs, outputs and intended goals that we would wish to have as a prelude to controlled trials (White & Stirling 2013).

Technological change offers scope for sophisticated and data-intensive monitoring of behavioural shifts and comparison with self-reporting. For example, social media, smart meters, intelligent domestic sensors, biometric sensors and ‘big data’ tracking of people in everyday practices (such as footfall in particular places) all provide opportunities for data gathering, modelling and measurement of behavioural patterns. These approaches all come together in concepts of the ‘smart city’ strategy for use of advanced ICT and integrated data analysis to manage urban systems. However, they all raise complex questions about consent, public engagement and democratic dimensions of use of new sensor and data management technologies. Smedley (2013) emphasises this point in relation to current debates over ‘smart cities’:

> *“Centralised operations systems must engage with citizens not simply monitor them; citizen groups must question policy and the use of big data, while also contributing to it; smart sensors in streets are still needed, as are those we choose to put into our houses. An holistic approach to smart city planning seems possible, but we are not there yet.”*

There are also limitations to be noted in the drive towards use of large-scale quantitative studies of behaviour change. RCTs and ‘big data’ modelling studies will generate a large amount of information, but on their own will not generate insight into the reasons, meanings and values attributed by citizens to particular changes and interventions. To gain insight in this respect, there will be a continuing need for in-depth action research and qualitative studies over time with citizens in the context of behaviour change initiatives (Warburton, 2008; White & Stirling, 2013; Dolan *et al*. 2010).

4. Trends: what can be learned from the way cities have developed; how might they change in future?

4.1 Historical trends

The UK is arguably home to the world's oldest industrialised urban areas, with London being the first city to reach a population of one million people in the early nineteenth century. Although the rate of growth has now slowed and the UK's cities are characterised by relatively mature infrastructures, a review of past transitions can be instructive for the future. We focus on the transitions in three major infrastructures that influenced the city's urban metabolism: its energy systems, transport systems, and water systems.

4.1.1 Energy

London was originally powered by locally available biomass and animal wastes, with an estimated 1.5 m^3 of wood (~17 GJ) required per person per year for heating and cooking needs in the 13th century (Galloway *et al*. 1996). However as London grew in the 15th century with income from the wool trade, energy demands increased and, by 1550 wood was being imported from over 40 miles away rather than the immediate hinterland; this led to a two-to-threefold increase in cost (Allen 2010). Coal, imported from Newcastle and surrounds, became increasingly popular and soon hearths and industrial processes were converted to use this fuel. Similarly the scarcity of animal fats for lighting purposes led to the development of town gas and then electricity. However these electricity and gas networks were highly localised. It was not until the 1930s that London became connected into national energy grids, enabling it to consume resources sourced from afar with almost no local indications of origin or indeed environmental impact. Overall these energy transitions can be summarised as showing "increasing efficiency under constraints" and marked by increasing energy efficiency of service provision, increasing per capita energy use, increasing complexity in the system's structure, and requiring a strategic overview to guide innovation (Rutter & Keirstead 2012).

A few studies have presented time series data for energy use in international cities. Kim & Barles (2012) provide a record of energy consumption in Paris from 1730 to 2000, showing particularly rapid increases in per capita energy use from 1950 to 1990. Similar findings are shown by Reiter & Marique (2012) for Liège, Belgium, with data since 1850. Baynes & Bai (2012) analyse the doubling of energy consumption in Melbourne from 1973 to 2005. Bristow & Kennedy (2013a) fit a non-linear model to data for Hong Kong and Singapore from 1965 to 2010, distinguishing between energy used to grow versus maintain the urban economies. There is also a substantial body of literature on historical trends in energy and material flows at national scales (e.g. Grubler 2012, Bernadini & Galli 1993).

4.1.2 Transport

The Industrial Revolution also transformed urban transportation, with knock-on impacts for urban metabolism. London's first inter-city terminus, Euston Station, opened in 1837, at the start of a rapid, often speculative, expansion of the UK's rail network known as "Railway Mania". In 1862, the London Underground opened revolutionising transportation within the city. Both of these infrastructures shaped the urban metabolism of London and its surrounds, enabling residents to escape the crowded boroughs of the city and commute in from nearby market towns. Goods could also be imported at low cost from the workshops of the Midlands and the North, helping to establish patterns of consumption that are familiar today. It is interesting to note however that the laissez-faire development of Britain's railways and London Underground, while creating densities of public transport access that are now seen as key to urban sustainability, also led to wasteful duplication and "enormous inefficiency" that a recent analysis suggests could have been avoided with a strategic integrated transport plan (Casson 2009).

4.1.3 Water

Two significant transformations to London's urban metabolism came with the introduction of water supply and wastewater disposal networks. Originally Londoners sourced their fresh water from springs, wells, and the Thames. As the city grew, these sources became increasingly polluted and private water companies began to compete to provide clean supplies, an arrangement that continued despite cholera outbreaks that in cities like New York led to municipalisation. In 1852 these private companies were regulated, setting standards for continuous access and cleanliness, but it was not until 1902 that the water networks were taken over by the Municipal Water Board (Salzman 2006). Public ownership continued until privatisation in 1989. The current operator now sources water from 102 water treatment works throughout the Thames Valley and provides a daily average of 2.5 billion litres of clean water to 9 million customers (Thames Water 2013a).

A similar story can be seen in the wastewater sector. Medieval London disposed of its wastes in cesspools, roadside ditches, or latrines constructed over water bodies. As the city grew, this led to unacceptable health risks and nuisance pollution (Hansen 2003). With £3.5 million in funding from Parliament, the first elements of London's modern sewer system were opened in 1865 to the design of Joseph Bazalgette (Cook 2001). The network has come under increasing pressure in recent years, due to population growth and development that reduces the permeability of the city's surface. This means that during high rain fall events (about 50 times per year) the sewers can discharge combined overflows of both wastewater and run-off directly into the Thames, violating European water quality standards. Operator Thames Water has recently been arguing for the construction of a new extension to the network, the Thames Tideway Tunnel, to cope with these events as well as the effects of future climate change and population growth, but the project remains controversial and is unlikely to be completed until 2023.

There are also significant interactions between water, energy and other infrastructures. Evidence suggests that integrating these networks can bring economic and environmental benefits, although the distribution of these gains may be unequal between infrastructure operators (Chertow & Lombardi, 2005).

4.1.4 Summary

While each of these infrastructures is unique, there are clearly some common themes. The first is that changes in London's socio-economic landscape were vital for driving technical transitions and associated material and resource flows. Booming economies, new scientific discoveries, and the threat of conflict all helped to promote new system configurations. A second issue is the reciprocal nature of many of these changes. The growth of London's railways in the 1830s made it easier for people to commute from the Home Counties; this meant people began to live in larger homes and changed their consumption patterns consuming more energy and consumer goods. Understanding these patterns of infrastructure-induced consumption, or *autonomous consumption*, are vital to assessing a city's vulnerabilities to resource supply constraints and its overall sustainability (Kennedy 2011). Third, the history of the electricity, gas, and railway networks highlight a tension between the enthusiasm and vision of private investors with the need for strategic oversight to avoid wasteful investment. Finally, each historical pattern of consumption reached some form of constraint, be it resource availability, environmental impact, cost, or other factors. In his study of the collapse of civilisations, Joseph Tainter notes that such constraints are typically overcome by new organisational patterns and information flows, but that these in turn require new resource flows to operate (Tainter 1988). Collapse ultimately happens when the speed of innovation needed to overcome these obstacles fails to keep pace. Other recent work on urban systems suggests the necessary pace of innovation is constantly accelerating, implying significant challenges for future cities (Bettencourt *et al*. 2007).

4.2 Lessons from new city developments

During the 20th century there were attempts to plan cities in new ways, such as Brasilia, New Delhi, or, in a UK context, Milton Keynes, and Ebenezer Howard's earlier garden cities (e.g. Letchworth in Hertfordshire, England, initiated in 1898); however, little can be said about the metabolism of these cases. Brazilian studies of urban metabolism have been conducted for Sao Paulo and Rio de Janeiro, which have low per capita GHG emissions due to favourable climates, high density, and low carbon electricity sources, but the current authors are not aware of studies of Brasilia. The metabolism of New Delhi is currently being analysed in a study of 27 megacities; preliminary results show that its metabolism is similar to the other large Indian cities of Mumbai and Kolkata. There have also been studies of various aspects of metabolism in other Asian cities, which have rapidly developed in recent decades (e.g., Schulz 2007; Pachauri & Jiang 2008; Dhakal 2009; Jago-on *et al*. 2009; Tanikawa & Hashimoto 2009; Yamasue *et al*. 2009. Tian *et al*. 2013; Zhang *et al*. 2014). Lessons on how to develop more sustainable urban metabolism can be drawn from the city-scale data presented so far (Figures 2-3 & 3-1), but further insights can be seen from new neighbourhood scale developments.

At the neighbourhood scale, there has been detailed examination of four low carbon community developments in Europe by Fraker (2013). Out of several possible sites, Fraker selected the communities of Hammarby (in Stockholm), Bo01 (Malmo), Kronsberg (Hannover) and Vauban (Freiburg) for several reasons: all four are large developments with at least 1,000 housing units (Table 4-1); they include commercial space as well as residential (at least 30% jobs to housing); aggressive approaches to designing sustainable metabolism were pursued, including transportation; and a sufficient amount of long-term performance data is available. (Fraker considered the BedZed site in the UK to be too small.) The planning of the sites, which started in the mid to late 1990s, also has some commonalities: all received substantial EU seed funding; three were developed in the context of international expositions or events; and all had masterplans designed by outside consultants, typically through competition. Local governments played strong roles: acting as master developers, assuming financial risks, and using their legal authority to direct responsible utilities to participate in integrated planning.

Table 4-1. Comparison of Four Sustainable Neighbourhood Developments (based on Fraker, 2013)

a) General design characteristics

	Hammarby, Stockholm	Bo01, Malmo	Kronsberg, Hannover	Vauban, Freiburg
Area (hectares)	200	22	70	34
Population (projected)	25,000	2,352	6,600	5,000-6,000
No. of units	11,000	1,567	3,000	
Density (population/ hectare)	125	107	94	147-176
Density (dwelling units/ hectare)	54	71	42	53
Jobs/Housing Ratio	47%	275%	67%	33%
Auto mode share	20%	50% (preliminary estimate)	Goal of approx 20% (no survey)	10-15%

b) Ground coverage

Coverage	Hammarby, Stockholm	Bo01, Malmo	Kronsberg, Hannover	Vauban, Freiburg
Buildings	15%	21%	18%	19%
Roads & parking	8%	9%	16%	11%
Green space	45%	32%	64%	68%
Water	22%	28%	2%	2%

c) Energy supply and demand

Energy	Hammarby, Stockholm	Bo01, Malmo	Kronsberg, Hannover	Vauban, Freiburg
Goal (kWh/m^2)	105	105	75	75
Measured (kWh/m^2)	154	167	101	56
	+49%	+59%	+35%	-25%
% renewable	22	100	52	80-90

The main lessons from these four European sites are as follows:

- Automobile mode shares of total transportation trips of 10 to 20% were achieved at Hammarby and Vauban. Traffic surveys are not reported for the other two sites, but auto mode shares are expected to be at or below the European average of 50%. All sites were designed to strongly promote sustainable transportation modes: weatherproof transit stops with real time information are located within 300-400 m of residences; transit vehicles have headways of 6-8 minutes; there are fine-grained street and block patterns with frequent pedestrian shortcuts, dedicated bike paths and bike parking facilities; traffic calming measures are in place and parking is limited to 0.2-0.8 spaces per housing unit.

- The ratio of hardscape to landscape is noted to be "radically different from that in the traditional city". The areas of hard-space (building footprint and surfaces for transportation) total no more than 30 to 40% on each site, i.e., green-space and water total 60% to 70%. This is achieved by having only 8-16% of space dedicated to roads and parking, and having moderate urban densities – averaging 80-100 units/acre – arranged in perimeter block style, i.e. with buildings fronting to streets with green-space behind. Such high amounts of green space add to the attractiveness of the four neighbourhoods. So resource efficiency can be achieved at medium densities and with room for green space.

- The average building direct energy demand for the four neighbourhoods is ~125 $kWh/m^2/yr$. This is approximately half that of standards at the time of design, although the Swedish standard today is 45 $kWh/m^2/yr$. All four sites employed high standards of building envelope design (insulation, windows, low air infiltration) and aimed for good quality construction as necessary first steps in reducing heating demands. Less attention was given to use of electricity; the three neighbourhoods with usage data exceeded their electricity targets.

- Vauban had the lowest energy demand of 56 $kWh/m^2/yr$, which was 25% below its target. This was achieved using passive solar design, natural ventilation, removable shading, and strong resident engagement. Kronsberg met its heating target by providing advanced training to contractors, detailed standards, careful inspection, blower door tests, and resident education. Bo01 and Hammarby missed their heating demand targets due to “flaws in construction, excessive glazing, orientation of buildings, and user behaviour.”

- All four neighbourhoods had notable advances in the use of renewable energy supply. Despite having the highest demand (167 $kWh/m^2/yr$), Bo01 obtains 100% of its energy from renewable sources: wind and geothermal (ground and seawater heat pumps plus solar). Vauban meets 80-90% of its energy demands through renewable sources: solar PV and waste-energy cogeneration burning wood chips. Kronsberg meets over 50% of its demand through wind and limited amounts of solar with the rest provided by a natural gas fed cogeneration plant. Hammarby obtains 22% of its energy (approximately 30 $kWh/m^2/yr$) from renewable sources, all from waste-to-energy systems.

Another sustainable site development at Masdar, near Abu Dhabi UAE, contrasts with those in Germany and Sweden, due to its considerably different climate. Masdar City is planned to be a net zero community of about 30,000 people. As of 2013, it comprised about 20 buildings, primarily housing the Masdar Institute – a research centre – accessed by an experimental personal transit system. The strategy for reducing energy demands in the community plan is to place buildings close together – using shade to reduce building cooling loads. Buildings occupy 50% of the ground coverage (versus 15%-21% in the European sites), with narrow streets, public spaces and court yards arranged similar to traditional Arab cities. Population densities will likely be slightly higher than at the European sites, reaching up to 250 persons/hectare. Energy demands in the city plans are expected to be over 200 kWh/m^2/yr, with about 75% of energy used for cooling. Electricity generated from on-site renewable sources is sufficient for the site to be carbon neutral with respect to local energy consumption.

4.3 Technological change

As alluded to in the previous discussion (Section 3.4) on behaviour change (which is addressed again in Section 4.4), the history of cities such as London clearly shows that technological change can lead to transformations of urban metabolisms. While technological forecasting is a highly speculative activity, it can still be instructive to think about how new innovations might shape urban metabolism in the future. Here we consider a selection of near-term (i.e. to 2030) technologies, infrastructures, and trends that might affect the urban metabolism of UK cities. Brief details on each innovation are presented below and for each we estimate its *impact* (i.e. will the technology lead to increased or decreased material and energy flows) and *likelihood of adoption*. Other innovations might affect the environmental impacts associated with urban metabolic flows, e.g. decarbonisation of grid electricity, but here we have considered only the material and energy flows themselves. The results are summarised in Figure 4-1 and discussed in the following sections.

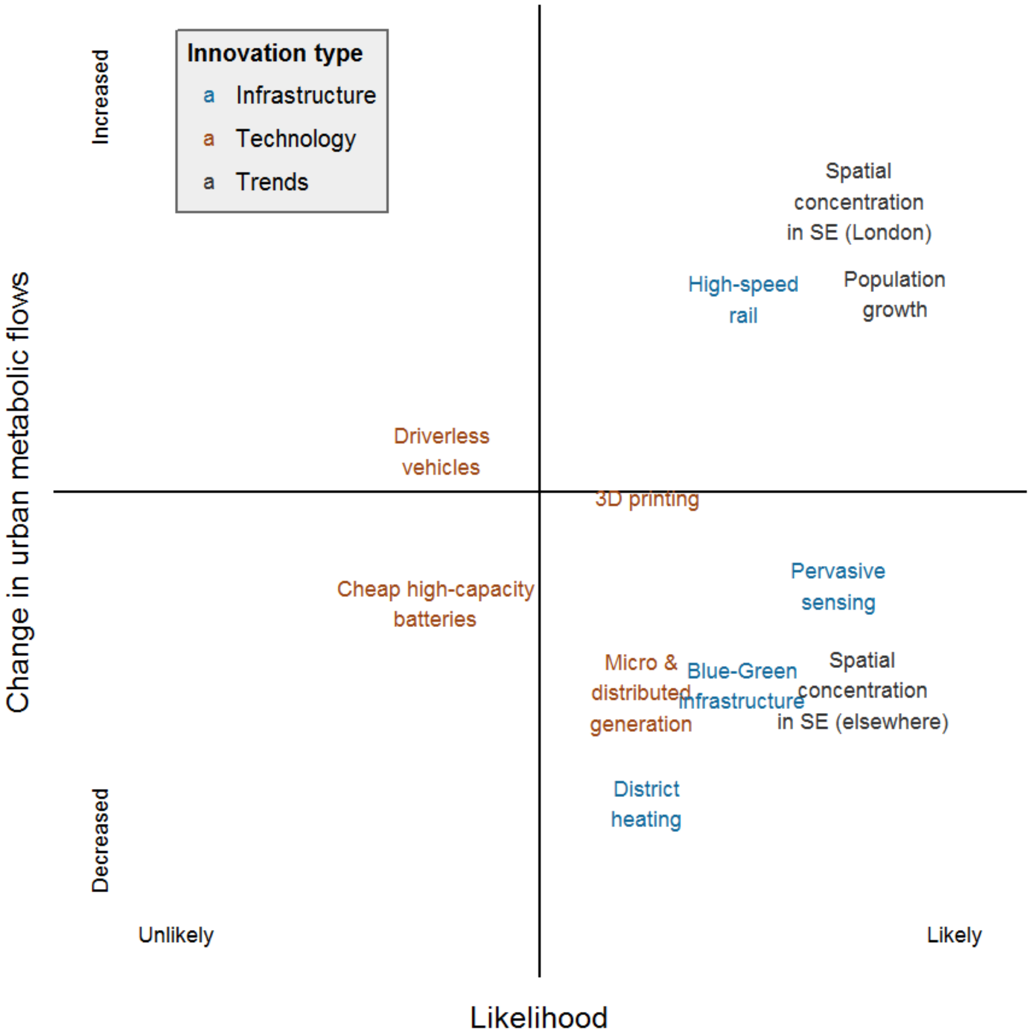

Figure 4-1. Estimated likelihood and impact of selected innovations on metabolic flows in UK cities

4.3.1 Technologies

- **3D printing.** 3D printing (or "additive manufacturing") technologies are already widely available, although the extent of their use within the economy remains very uncertain in relation to impact, scale-up potential and regulatory issues. A US Department of Energy study suggests that the technology could reduce energy costs by 50% and material needs and costs by 90% (US DoE 2012). However, other reviews suggest that depending on the printing process, there may be substantially increased energy consumption requirements per unit of material processed (Huang *et al*. 2012). The raw materials for printing must also be transported to the city and, as with the oft-promised "paperless office", there may in fact be an increase in object creation.

- **Cheap high-capacity batteries.** Low-cost high-capacity electrical storage could be a valuable technology for facilitating the use of intermittent generation sources in electrical grids and for electric mobility. A range of technologies are currently being investigated to provide storage from hour to day time scales with capacities up to 1 GWh (Energy Research Partnership 2011). These could help to improve the efficiency of system operation, assist the electrification of heat and transport, and reduce the magnitude of cross-border energetic flows.

- **Micro- and distributed generation.** A range of established technologies exist for electricity generation within the city (< 50 kW for microgeneration, up to MW scale for distributed units). Depending on the prime mover (e.g. local versus imported biomass, cross-boundary wind or solar flows), these technologies could enable better harvesting of available resources within the city but the likely generation, particularly from renewable sources, would be much smaller than demands. However combined heat and power technologies (see District heating below) would enable a greater fraction of the energy in imported fuels to be used (e.g. up to 90-95% efficiency for some large plants) and could use locally generated municipal solid waste. These technologies are already well established throughout Scandinavia.

- **Driverless vehicles.** Small demonstration projects already exist (e.g. Heathrow airport) and these autonomous "pods" are also an integral part of the Masdar low carbon city plan in Abu Dhabi. Widespread adoption is more speculative, but could lead to increased leisure/work time, resulting in more income or consumption. The impact on resource consumption would depend on the vehicle technology but there could be an increase in energy consumption if city-dwellers switch from public transportation or begin to use these pods for short-trips which are currently completed by walking or cycling.

4.3.2 Infrastructures

- **District heating.** District heating networks are vital urban infrastructures in order to capture the benefits of combined heat and power. The city of Gothenburg, Sweden, for example have nearly 1000 km of infrastructure allowing it to capture over 90% of the useful energy in imported fuels. Recent projects, for example in Southampton and the Bunhill scheme in Islington, are beginning to demonstrate the benefits here in the UK.

- **Pervasive sensing.** Pervasive wireless sensor networks enable infrastructure operators to operate systems more efficiently through real-time monitoring and control. While the impact on urban metabolic flows may be relatively minor, a significant benefit would be the increased availability of data enabling governments, private firms, and analysts to better understand spatial and temporal patterns of resource consumption and its effects, thus opening the door to potentially transformational innovations.

- **High speed rail.** The proposed HS2 high-speed rail link would have a relatively minor direct effect on urban metabolic flows, as most of the energy and infrastructure would be required in the countryside between cities. (Of course the users of this infrastructure are likely to be urban dwellers thus increasing energy consumption and emissions when measured from a consumption perspective.) However if the system achieves its economic goals there could be substantial increases in urban metabolism due to the redevelopment of the station areas (e.g. Curzon Street and Euston), new commuting patterns, and general economic growth as seen in China (Zheng & Kahn 2013).

- **Blue-Green Infrastructure.** Blue-Green Infrastructure, which uses vegetation to improve urban water management (e.g. storm run-off), has the potential to substantially alter the urban hydrological cycle. Although demonstration projects are just beginning in the UK[13], these infrastructures could reduce the need for engineered metabolic processes, such as water treatment works, the proposed Thames Tunnel, as well as providing space cooling.

4.3.3 Trends

- **Population growth.** The UK's population is expected to reach 71 million in 2030, due to both natural increase and inward migration. In London, the population is forecast to grow to 9.95 million by 2031. [14] However the population is also ageing, which may affect material and energy consumption due to increased leisure but also changes in income associated with retirement. Inequality is also increasing in the UK which will shape consumption trends. One important factor is household formation rates, as additional households require more buildings, appliances, and other forms of autonomous consumption.

- **Spatial concentration of economic activity.** A recent analysis of 2001-2011 census data shows internal migration patterns from urban north to urban south and a decline in urban to rural migration (Lomax *et al*. 2014). This suggests that urban areas elsewhere may see net outflows (e.g. Birmingham[15]) and is consistent with observations of an economic recovery in the South-East but less so elsewhere. The expectation is that economic activity will continue to grow in London and the South-East owing to agglomeration effects, thus increasing metabolic flows in London (energy, water, but also materials for related infrastructure). See Section 1.5 for more detail.

- **Technological change.** From an action oriented perspective, different infrastructure or technology strategies can be taken by cities to reduce their GHG emissions depending on city characteristics (Kennedy *et al*. 2014). Climate clearly determines the extent to which building retrofit and new building approaches are important (Figure 4-2). Electrification strategies, e.g., using electric vehicles or heat pumps, are carbon competitive for cities served by electric grids with intensities below approximately 600 tCO_2e/GWh (Figure 4-2). The use of district heating systems and heavy rapid transit (i.e., subway or regional rail) are broadly more economically viable for cities with higher urban population densities (over

[13] www.climate-kic.org/projects/blue-green-dream/
[14] www.london.gov.uk/sites/default/files/Update%20052013%20GLA%202012%20round%20population%20projections.pdf
[15] www.neighbourhood.statistics.gov.uk/HTMLDocs/dvc25/

approximately 6,000 persons/km^2). Installation of building-integrated photovoltaics (BIPV) is a key strategy for cities that are both low density and currently served by high carbon electricity grids. If suitable polices, financial or otherwise, are put in place to encourage low carbon urban development, then future technological change in cities may develop along the lines broadly sketched out in Figure 4-2.

The rate at which low carbon urban transformation can occur depends upon the rates of diffusion of low carbon technologies and turnover of the building and vehicle stocks. Mohareb & Kennedy (2012, 2014) have explicitly modelled the (direct) GHG implications of rates of turnover in the building and vehicle stocks in the Greater Toronto Area (Figure 4-3). Direct GHG emissions for the Toronto region are already on a downward trend due to the phase out of coal combustion from the Ontario electricity grid. Even lower emissions could be achieved through transformation of the road vehicles, e.g., to electric vehicles, which could be achieved within a period of about 15 years (i.e. the lifetime of an automobile). In the long-run one of the greatest barriers to achieving deep emissions cuts is the building stock, which take longer time periods to retrofit or replace.

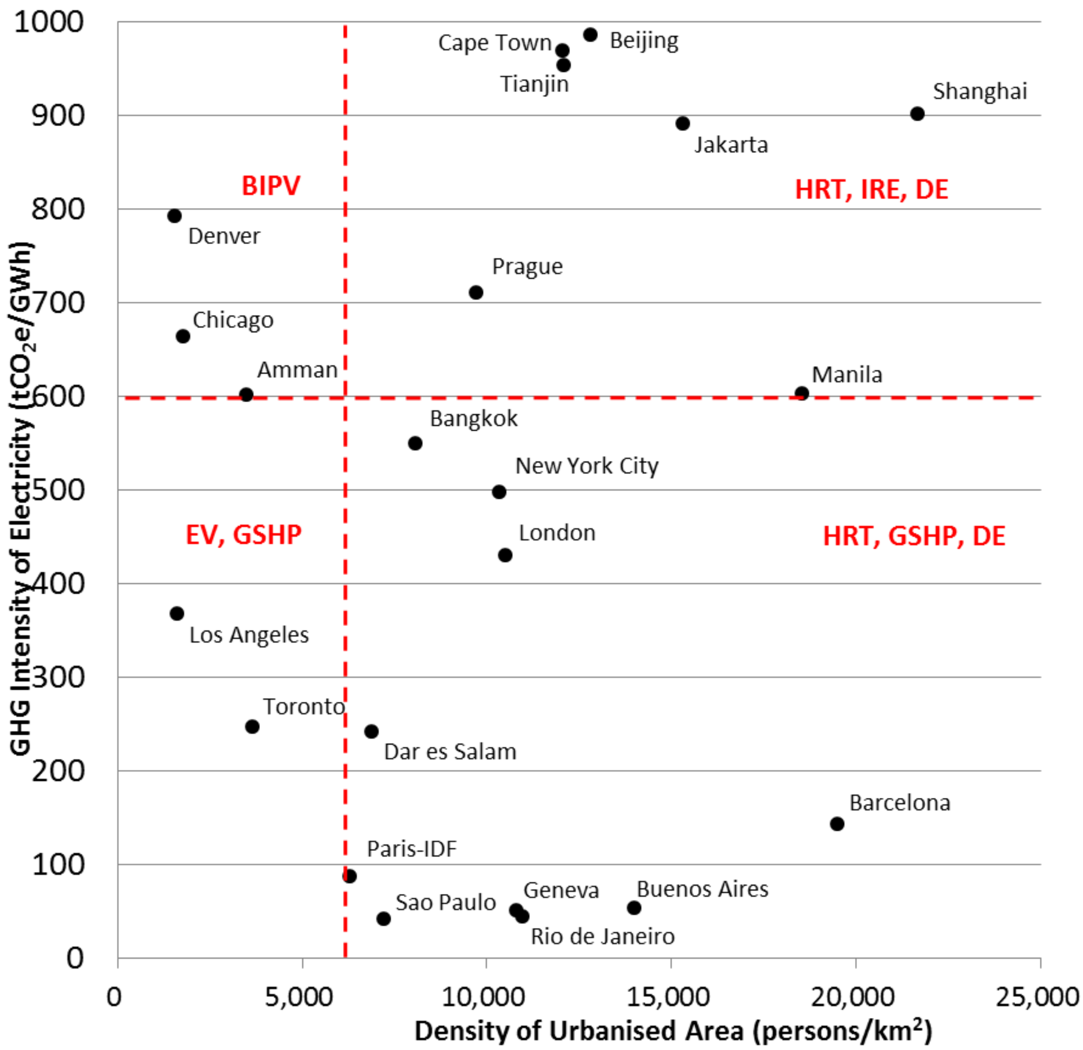

Figure 4-2. Examples of low carbon infrastructure strategies tailored to different cities. Prioritisation according to urban population density and the average GHG intensity of existing electricity supply. (Figure 4 from Kennedy *et al*. 2014). BIPV: Building Integrated Photovoltaics; DE: District Energy; EV: Electric Vehicles; GSHP: Ground Source Heat Pumps; HRT: Heavy Rapid Transit; IRE: Import Renewable Electricity

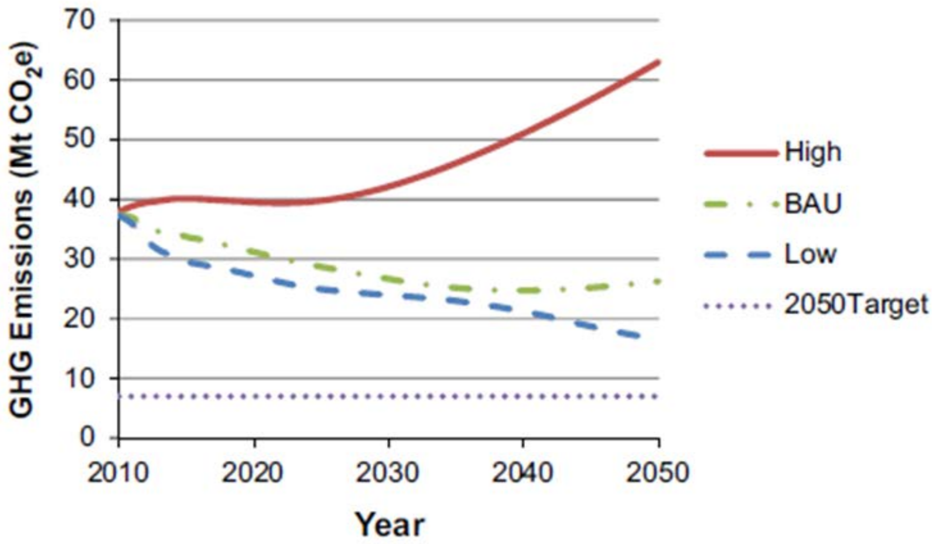

Figure 4-3. Modelled GHG emissions for the Greater Toronto Region under low, high and business as usual (BAU) scenarios (Figure 4 from Mohareb & Kennedy, 2014)

4.4 Behavioural change

Although cities impose major demands on resources and environments, as discussed above, and in some cases seem to be at the limits of effective governance (Bettencourt *et al*. 2007), they are increasingly recognised as potential hubs for innovation in sustainable consumption and production (Barber 2013; Birkeland 2009; Brand 2010). Cities have been at the forefront of policy action on climate change, waste reduction and resource efficiency, and networks of mayors and other urban leaders have become a major driving force for local sustainability innovations (see www.iclei.org ; Barber, 2013).

Below we briefly note a number of areas in which urban trends intersect with behavioural and attitudinal changes that can support sustainable development:

> Ever since the Rio 'Earth Summit' of 1992 there has been significant activity among local and regional governments and local stakeholders in promoting sustainable development (see www.iclei.org) and in recent years there has been a clear trend for city governments to take the lead over national ones in planning and acting on unsustainable development (Barber, 2013), on the basis that national action is often perceived to be inadequate, that cities are sensing their vulnerability to ecological risks such as sea-level rise, floods and increased storms on coasts, and that it is easier for mayors and city governments to secure consensus and resources for action than it is for national governments which face great challenges in aggregating political interests and mobilising actors. It is likely that further integration of activities by city leaders on sustainable living and strategies for sustainable development will take place, possibly attempting to by-pass national administrations if these are seen by city leaders as failing to take serious action on climate disruption and transition to a sustainable energy system. Relevant initiatives include the ICLEI Climate Network, now covering some 700 cities (www.iclei.org) ; the Global Town Hall event held by mayors and other city leaders at the Rio + 20 Summit in 2012; the R20 network of regional governments, an initiative focusing on Green Economy strategies at sub-national level (www.regions20.org); the EU's Energy Cities initiative (www.energy-cities.eu); and the C40 network of cities, a Climate

Leadership Group of mayors and other city leaders committed to local action to mitigate and adapt to climate change (www.c40.org).

Cities worldwide have large populations of young people: some 70% of those under 25 in 2050 are expected to be living in cities (UNICEF, 2012; Chawla, 2002), and so understanding the consumption patterns, attitudes, values and potential for behavioural change among the urban young is an important challenge for policy-makers and researchers concerned with urban metabolism models and strategies for sustainable city development. There are interesting signals of generational shifts in consumption: see for instance the trend away from car ownership and use among urban young people in the USA (Davis & Dutzik, 2012; Davis & Baxandall, 2013). The CYCLES project of UNEP and a range of partners including the University of Surrey will generate a global survey series on the consumption experiences and life expectations of young people aged 12 to 24 in 21 cities at five-year intervals (Abbas *et al*., 2013). This kind of research should prove to be a valuable resource for policy-makers, academics and modellers.

Cities worldwide, and not only in the Global South, are also the focus for large-scale action by religious organisations and communities. This is a period of major expansion in urban faith communities, notably in cities in Africa, Asia and Latin America, but also in European cities (Micklethwait & Wooldridge, 2009). This phenomenon is relevant to the discussion above on behavioural change. Faith communities are significant for sustainable development in two respects: first, there are increasing efforts to integrate pro-environmental awareness and action into the large-scale social and economic services and investments provided by churches and faith communities (see www.arcworld.org); and second, churches and religious communities are important networks for transmission of ideas, practices and incentives for their members, and so have the potential to be harnessed for promotion of sustainable lifestyles in the context of faith traditions and teachings.

The extreme complexity of urban environments and societies makes the application of complex systems theory to cities highly relevant. Recent work by the Santa Fe Institute highlights the potential insights to be gained from such research on urban metabolism, with a particular focus on agglomeration and scaling effects in cities (Bettencourt 2013; Bettencourt *et al*. 2007). These studies suggest that scaling effects generate ever more connectivity between citizens and organisations as city size increases, and that continuous innovation is required to enable economic growth and prevent collapses. This work underlines a crucial point about cities in relation to sustainable development: they can be seen both as exemplary cases of unsustainability, for example in relation to inequalities and health risks (Rayner & Lang, 2012); and also as potential solutions to unsustainable development, enabling concentration of people and innovation, resource efficiencies, ‘smart city’ technological systems, and tight feedback loops of connectivity, imitation, adaptation and incentive effects (Brand, 2010; Birkeland, 2009; Provoost, 2013).

Cities are already the locus of many ‘smart technology’ testbeds. Given the trends identified above, we can expect cities also to be increasingly the focus for initiatives on sustainable design and development of housing, food production, mass transit and localised energy systems (Birkeland, 2009). These will all be developed within a framework of urban sustainability that aims to reduce food and energy insecurity, increase resilience to climate disruption, manage city growth, and reduce pollution (e.g. via more mass transit and remote working) and waste. Opportunities in the UK seem likely to be focused on cities with high levels of ambition for sustainability (London, Bristol) and on the proposed new generation of garden cities (Hall & Ward, 1998; TCPA, 2012; TCPA, 2014).

Cities have the potential to be the leading laboratories for sustainable behaviour change, but this will not be realised automatically. Density of networks and feedbacks in ‘urban metabolisms’ mean that unsustainable practices can be reinforced and embedded just as readily. Moreover, cities feature many infrastructural legacies that generate ‘lock-in’ to existing practices for citizens, making behavioural change hard to achieve. Accordingly, behavioural change for urban sustainability will need to be based on a rich mix of policies that deal with structural blockages and disincentives and also with ‘downstream’ local opportunities based on creating the social context for positive behaviour change in households and communities.

4.5 Resilience and vulnerability

The energy stored in cities has been used as one measure of the resilience of cities to energy supply shocks. While urban metabolism is dominated by flows of energy in and out, there are small quantities of energy stored in cities, which become important when cities are subject to shocks especially relating to energy supply. Forms of stored energy include, for example, petrol in vehicle tanks, food in supermarkets and homes; and electric battery storage. Another possible form is underground thermal energy storage used for seasonal heating and/or cooling. A study of Greater Toronto by Bristow & Kennedy (2013b) estimated that there are 6 days’ stock of petrol in the urban region, primarily stored in vehicle tanks; and 12 days’ stock of diesel. These residence times are based on normal usage rates, which potentially could change under states of stress. The same study estimated that there were 20 days stock of food in the Toronto region, most of it stored in supermarkets; some of this stock may be lost to spoilage in event of a disruption to energy systems. This measure of urban resilience is, however, limited; it does not address, for example, self-sufficiency in water and food – or the social capital and social infrastructure that enable a city to be resilient in the face of upheavals such as race riots.

A recent international assessment report highlighted several connections between urban resource consumption and a changing climate (Rosenzweig *et al*. 2011). Depending on the climate and the resource in question, resource demands may either increase or decrease; for example, in temperate regions, a warming climate may led to significant increases in electricity demand due to the adoption of air conditioning (Sailor & Pavlova 2003). Climate-related risks to urban metabolic infrastructures, such as energy and water supply systems, was highlighted as a common and major area of concern, particularly for coastal cities (Rosenzweig *et al*. 2011).

4.5.1 Complex systems and resilience

Resilience is the capacity of a system to retain and / or recover function after undergoing stresses and shocks. The concept has become a major element in debates about the nature of sustainable development in the face of seemingly inevitable disruptions in global and local ecosystems (see for example Blewitt & Tilbury, 2014). Urban resilience (Monaghan, 2012) concerns the capabilities of cities to meet major challenges of adaptation and recovery from unexpected events. For example:

- disaster recovery, for example from earthquakes, major floods, hurricanes and terrorism;
- economic and social adaptive capacity: the ability to maintain or restore good levels of employment and income following economic shocks and recessions, and to sustain or restore social capital in the wake of social stresses, such as a major influx of migrants or refugees, or the displacement of residents by environmental shocks (such as floods);

- long-range adaptive capacity: this covers the ability of city governors to plan for foreseeable technological, economic, social and ecological challenges and to maintain or enhance the city's capacities to function as a prosperous economy and a socially cohesive entity. Long-range adaptation challenges include: risks from climate disruption (especially for coastal cities – see Bowman *et al*. 2008; Douglas and Leatherman 2008); competition for investment and need to maintain or improve the attractiveness of a city's amenities; need to update ICT infrastructures for 'smart city' development; demographic and cultural tensions from large-scale migration and from multi-ethnic mixes in cities; the challenges of food security and energy security in a world of climate disruption (Vogt *et al*. 2010; Hopkins 2008; Worldwatch Institute 2007). The long-range aspect of resilience also includes a literature on the scope for moving from a defensive and recovery-based view of resilience to one that sees cities as dynamic 'regenerative systems' that can use the challenges noted above to create new forms of prosperity based on ecologically sustainable urban environments (Girardet, 2012; Monaghan, 2012).

Managing urban resilience is a complex issue and raises difficult questions about governance and participation of citizens in shaping a sustainable future (Blewitt & Tilbury, 2014). Cities as complex systems have advantages and disadvantages in relation to resilience and recovery from stresses and shocks. The potential advantages are:

- redundancy: highly diverse socio-economic niches and ability to cope with the loss of, or damage to, specific places and organisations;
- diversity: multiplicity of skills, knowledge and locations of expertise;
- ingenuity: capacity for adaptation, entrepreneurship, innovation, rapid learning, imitation and feedbacks, density of support networks for households and organisations;
- mobilisation of internal and external resources: major cities can call on large-scale financial and human resources, and, given their strategic importance, on sources of external aid.

The potential disadvantages of cities as complex 'metabolisms' in relation to stresses and shocks include:

- high dependence on imported food and energy, with relatively low capacity for local provisioning (e.g. urban gardens and local community energy systems);
- vulnerability of city centres to natural and man-made hazards, such as floods, storm surges and sea-level rise (see Bowman *et al*., 2008, on New York), and concentration of major organisations and assets in central districts;
- concentration of populations and vulnerability to spread of diseases;
- urban bottlenecks in transportation and movement of people and goods in the event of systemic shocks;
- feedbacks of disadvantage as socio-economic problems in urban cores generate a spiral of decline, as has happened notoriously in the case of Detroit.

Much depends on the wider socio-economic condition of the city in question and its hinterland and national context. Rich cities have clear advantages over poor ones. But there are vulnerabilities for rich urban areas and advantages for poor ones. For rich cities, normally well buffered against physical and economic shocks, major stresses and incidents could come as a massive blow from which recovery is hard to manage. In poor cities, where everyday hardship is the norm for the majority and shocks from natural disasters are relatively common, there may well be great resourcefulness and grassroots resilience. Brand (2010) offers an optimistic view of the ecological sustainability and general resilience and ingenuity in 'squatter cities' with extensive slums and informal economies.

Improving resilience in cities is closely bound up with richer understanding of urban metabolism and with better modelling of the flows of energy and materials and people in urban systems. However, it is also about richer understanding of behavioural change – see section 3.4 above – and the ways in which citizens can collaborate and contribute to governance for urban sustainability (Blewitt & Tilbury, 2014; Girardet, 2012; Monaghan, 2012).

4.5.2 Equity

In relation to urban metabolism, social equity refers to the complex systems of socio-economic differentiation at work in cities, the extent of income and wealth inequalities, and of differences in capacities of citizens to participate in the local economic communities and city governance. There is extensive evidence to show strong correlations between high inequalities (of wealth, income, health etc.) and low levels of trust and social cohesion (Wilkinson & Pickett, 2009). Cities worldwide are places where extremes of wealth and poverty co-exist – the 'super-rich' and the 'super-poor' (Dorling, 2013) – and also where extensive networks of dense social connectivity can help alleviate the disadvantages faced by the poor. In analysing the prospects for urban sustainable development, we need to bring into the assessment not only ecological sustainability and economic resilience and vitality, but also social equity: are divisions of wealth, income and well-being unjustifiably great and potentially dysfunctional for the city as a whole system?

What can be done to reduce unjustifiable and risky levels of inequality? One approach that is potentially useful in considering social equity in relation to urban metabolism is the 'capabilities approach' to welfare economics and ethics pioneered by Amartya Sen (1999). In this approach, policy should focus on the capabilities of citizens – their capacity to lead fulfilling and responsible lives: this requires that they have the ability and access to sufficient resources to be able to flourish. In relation to urban metabolism, this leads to a focus not so much on income and wealth differences *per se,* as on removing foundational barriers to flourishing, such as hunger, fuel poverty, illiteracy, poor education, poor access to livelihoods, barriers to setting up businesses and community initiatives, lack of core infrastructures (for mass transit, clean water affordable electricity) and so on. Beyond this, considerations of equity in relation to sustainable urban systems lead to exploration of the policies that can support grassroots innovations for local food-growing, community-based energy systems, mutual enterprise, and so on (Blewitt & Tilbury, 2014; Seyfang, 2009).

5. Recommendations: using urban metabolism for strategically advancing environmental sustainability

1. There are practical reasons for analysing urban metabolism and using it to develop tools for strategic support of sustainable development at national and local scales. The urban metabolism provides necessary data on energy use, industrial processes and wastes required when calculating GHG emissions for cities, as well as measures of resource use and efficiency. With such critical data, assessment tools and strategies (e.g., Figures 4-1 to 4-3) can be developed which guide urban and industrial development towards environmental sustainability. Cities where urban metabolism-like frameworks have been used in the integrated design of sustainable neighbourhoods have achieved substantial reductions in energy intensity compared to conventional designs. A strong example is Vauban (Freiburg Germany), which uses under 60 kWh/m^2 for building energy use.

2. Data on the greenhouse emissions for global cities shows that there is a relationship between the urban metabolism and the GDP of cities (Figure 5-1). There is, however, substantial variation in the data which can be explained by other factors, such as climate, urban design and carbon intensity of electricity. Even where a city has relatively low direct GHG emissions, it can still have a large carbon footprint due to the emissions associated with household consumption that occur beyond the city boundary.

3. The UK has sub-national energy statistics which can be used in urban metabolism studies and help local authorities aiming to meet National Indicator (NI)186 requirements (DCLG 2015). There have been a few studies of the urban metabolism of UK cities, such as Birmingham, Leeds, London and York, that have supported calculations of carbon footprints. Extensive carbon footprints for UK regions have also been determined from household consumption surveys. There have also been many ecological footprint studies of UK towns and cities, but these have poor scientific credibility.

4. It is important to take a life cycle (cradle to grave) perspective when assessing environmental impacts of cities, otherwise polices may shift and possibly magnify environmental burdens rather than reduce them. One comprehensive technique for assessing life cycle environmental impacts is through application of input-output models. These have been used, for example, to assess urban-scale direct and indirect water consumption throughout the UK.

5. The UK’s carbon footprint due to consumption has increased 10% since 1993, despite increases in efficiency of production. Fifteen percent of household expenditure is on imported goods, but they account for 40% of embodied carbon emissions. Changing patterns of consumption occur due to changes in population, socio-demographics and expenditure patterns; household income is the strongest predictor of consumption.

6. Future changes to urban infrastructure will be important for determining quality of life and for reducing environmental impacts. Quality of life depends more on the quality of the infrastructure than on consumption of luxury goods. People’s lifestyles are often locked-in by infrastructure systems and so even if they wanted to live more sustainably, it is often not possible. Infrastructure lock-in is a strong determinant of consumption (both social

norms but also what is needed to operate the infrastructure). Infrastructure systems are capital-intensive systems and last for decades.

7. The history of London has some important lessons on the development of infrastructure. To avoid wasteful investment a balance needs to be struck between the enthusiasm and vision of the private sector and the need for public oversight. Some strategic planning is needed to get the best results; infrastructure development should not just be left to the market.

8. Changes in the wider socio-economic environment often drive technological change and its associated material and resource flows. That said, the economics of future cities may be driven in part by technological change responsive to environmental/resource stresses. New organisational structures may be required so that innovation can keep pace in response to environmental challenges. Studies of urban metabolism – from an industrial ecology perspective – provide a good sense of which potential future urban technologies would be sustainable.

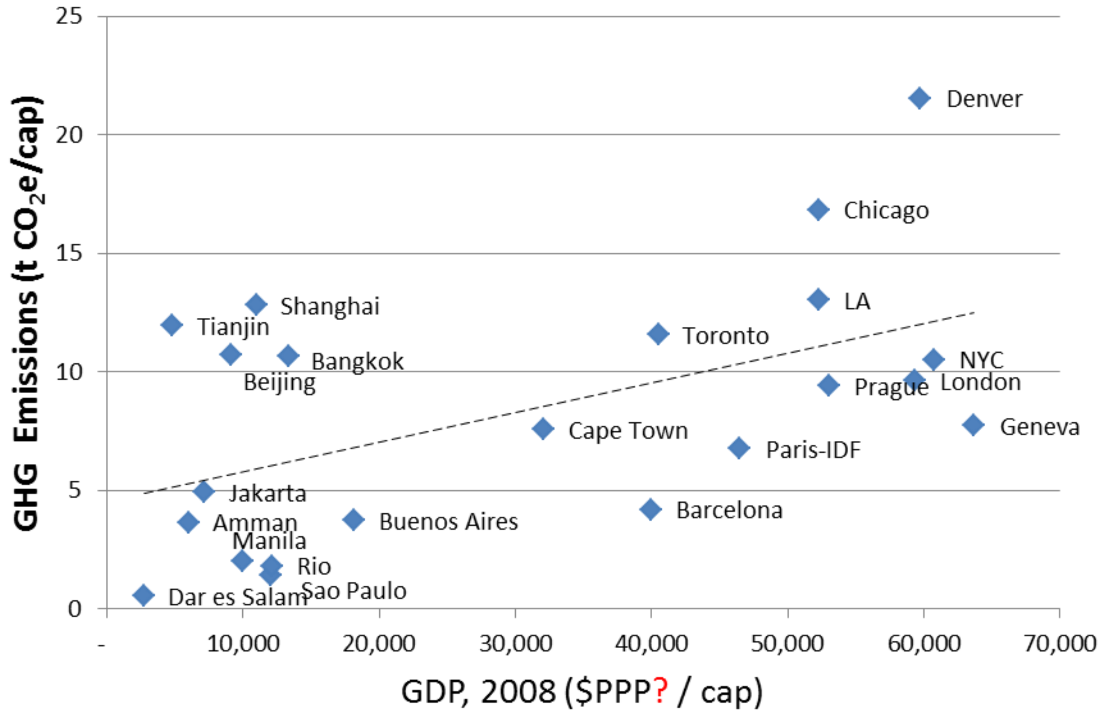

Figure 5-1. Direct per capita GHG emissions and per capita GDP (2008, $PPP) for a subset of global cities (Figure 2 from Kennedy *et al*. 2014)

Appendix: analysis of stocks and associated flows

To understand the importance of quality of a stock such as the built infrastructure in a city, it is convenient to use a formulation introduced by the IPCC in the 5th Assessment Report (2014)[16]. The GHG emissions (g) over a specified accounting period (for example, tonnes CO_2e per year) associated with a sector producing materials and products for stocks which deliver quantifiable services (such as person-km of transport during the accounting period) can be broken down in a form of Kaya (1990) relationship, showing the significance and interrelationship of the different terms contributing to GHG emissions, as:

$$g = (g/E) \times (E/p) \times (p/S) \times (S/d) \times d \qquad \text{(A-1)}$$

where E is the *energy used* (e.g. GJ per year) within the specified time period to produce a quantity p of materials and products (e.g. tonnes per year), S is the stock of durable products (e.g. tonnes) and d is the quantity of service delivered in the time period through use of those products.

This expression is conceptual, but it reveals the significance of the different terms:

g/E is the *emission intensity* of the sector expressed as a ratio to the energy used: the GHG emissions arise largely from energy use (directly from combusting fossil fuels, and indirectly through purchasing electricity and steam) and therefore depend most critically on the emission intensity of the background energy system of the economy where the goods in question are made. However, emissions also arise from industrial chemical reactions. In particular, producing cement, chemicals, and non-ferrous metals leads to release of significant “process emissions” regardless of energy supply.

E/p is the *energy intensity* of production: approximately three quarters of industrial energy use worldwide is required to create materials from ores, oil or biomass, with the remaining quarter used in the downstream manufacturing and construction sectors that convert materials to products (IPCC 2014). In some cases, particularly for metals, E/p can be reduced by production from re-used components or recycled material (see below), and can be further reduced by exchange of waste heat and/or by-products between sectors through the approach known as industrial symbiosis.

p/S is the *material intensity* of the sector: the amount of material required to create and maintain the stock. This depends on product or infrastructure design, including product life, re-use and recycling.

S/d is the *product-service intensity;* i.e. the stock required to deliver the required service. It can be reduced by using the products more intensively, for example through shared use of cars or appliances.

[16] A form of this equation is given in IPCC (2014) but the explanation given here differs from that in IPCC (2014).

Of the terms in eqn. A-1, p/S, S/d and d are relevant to the discussion of urban metabolism, whereas the others are related to background industrial production and energy systems. The importance of product life, re-use and recycling are illustrated by Figure A-1, in which p is the flow of goods or materials into stock (as in equation A-1) and q is the outflow of stock at the end of its (first) service life.

From the material balance,

$$dS/dt = p - q \qquad \text{(A-2)}$$

Of the flow q, a fraction f_1 is re-used while a fraction f_2 is recycled. The energy requirement per unit for re-use is e_1 and for recycling is e_2; normally e_1 is substantially smaller than e_2. The energy required for virgin material product is e_3, which is usually larger than e_1 or e_2; for steel for example, e_2/e_3 is in the range 0.4 to 0.5 whilst e_1/e_3 is in the range 0.1 to 0.2 (Allwood *et al*. 2010). The energy required to provide the flow of goods p to maintain the stock S (E in equation A-1) is then given by:

$$E = e_1f_1q + e_2f_2q + e_3[p - (f_1 + f_2)q]$$

which can be rearranged as:

$$E = e_3p - q[(e_3 - e_1)f_1 + (e_3 - e_2)f_2] \qquad \text{(A-3)}$$

Equation (A-3) shows the importance of maintaining high rates of re-use (f_1) as preferable to recycling (f_2) but also shows that the most significant way to reduce the environmental impacts of building up and maintaining stock is to minimise the material flows p and q.

To explore this further, we consider the case where the stock of goods in use changes little over time; i.e. dS/dt is negligible so that $p \approx q$ and

$$p = S/T \qquad \text{(A-4)}$$

where T is the average service life of the stock. Referring to equation (A-3), normal commercial pressures act to minimise the energy terms, but current business models rarely promote extension of service life so that there is more scope for extending T. Therefore the options for reducing energy use for most components of urban infrastructure and material products are, in priority order (Clift & Allwood 2011):

- Extend service life, T, to reduce p;
- Intensify use to reduce S;
- Increase the proportion of post-use product re-used, f_1;
- Increase the proportion of post-use product recycled, f_2;
- Reduce the energy required for recycling, e_2;
- Reduce the energy required for re-use, e_1;
- Reduce the energy required for primary material production, e_3.

The specific case of clothing (Section 1.5.2) provides an immediate illustration of the applicability of this analysis; short product life leads to high throughput. For the built environment, the analysis provides a basis for assessing the efficiency (or quality) of buildings and infrastructure (d/S) and also the benefit of retrofitting existing buildings and infrastructure (a form of re-use) and recycling components and materials from buildings demolished.

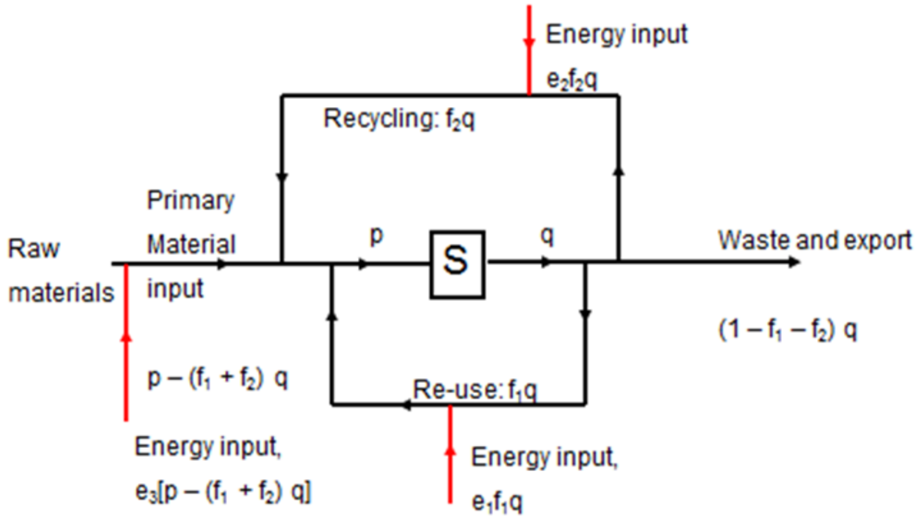

Figure A-1. Re-use and Recycling (after Clift & Allwood 2011)

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