SEVENTH FRAMEWORK PROGRAMME
THEME 6: Environment (including climate change)

Contract for: **Collaborative Project**

D.2.1
Inventory of current state of empirical and modeling knowledge of energy, water and carbon sinks, sources and fluxes

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Part 0: Introduction

0.1 Purpose of the document

This document is the D.2.1 - *Energy, Water and Carbon* - *State of the Science in Urban Areas*. The **aim of this document** is to review energy, water, carbon and air quality measurement and modelling for urban areas.

Globally, urban populations continue to grow. Current estimates predict that in Europe more than 75% of people will live in towns and cities by 2020; worldwide this will exceed 55% (UN 2007). Cities, and their inhabitants, are great consumers of water, energy, and carbon. The flows of water, energy and carbon into and out of cities, and their exchanges within, are key to the sustainable design of cities. While there have been significant advances in the understanding of these exchanges by natural scientists, this insight is not routinely used to inform sustainable urban planning processes or decisions.

The BRIDGE project, funded by the European Commission FP7, aims to develop a Decision Support Systems for urban planning end users, which draws on current scientific knowledge of the urban biophysical system, specifically energy, carbon and water exchanges, using the conceptual framework of urban metabolism. Examples are drawn from global cities, with a particular focus on European cities where data exist.

0.2 Document Structure

Chapter 0 is the introduction of the document (current chapter) which includes: the purpose of the document, the document's organization, the list of definitions and acronyms used in this document, the list of applicable and referenced documents and the BRIDGE project overview.

In this report an overview of the current status of knowledge of energy, water and carbon exchanges in cities is presented with an emphasis on uncertainties and needs for further research. The document is structured into **three substantive parts** – **water**, **energy** and **carbon**. Attention in each focuses on conceptual frameworks, scales (Part I addresses scales generally and this is not repeated in the other sections), methods of measurement (direct and through remote sensing) and modeling, and needs for further work. This is intended to provide context to the BRIDGE project.

0.3 Definitions and Acronyms

Notation and Acronyms for Part I

Symbol	Meaning
a_1, a_2, a_3	regression coefficients
c_p	specific heat capacity of air at constant pressure (J kg^{-1} K^{-1})
C_a	volumetric heat capacity of air (J m^{-3} K^{-1})
K_H	eddy conductivity (m^2 s^{-1})
K_V	eddy diffusivity for water vapor (m^2 s^{-1})
L_v	latent heat of vaporization (J kg^{-1})
$LW\uparrow$	outgoing longwave radiation (W m^{-2})
$LW\downarrow$	incoming longwave radiation (W m^{-2})
$Q*$	net all wave radiation (W m^{-2})
ΔQ_A	net advected heat flux (Q_{in} - Q_{out}) (W m^{-2})
Q_E	turbulent latent heat flux (W m^{-2})
Q_F	anthropogenic heat flux (W m^{-2})
Q_G	ground heat flux (W m^{-2})
Q_H	turbulent sensible heat flux (W m^{-2})
Q_{in}	advected flux into a volume (W m^{-2})
Q_{out}	advected flux out of a volume (W m^{-2})
ΔQ_S	net heat storage change within a control volume (W m^{-2})
S	other sources and sinks of the UEB (W m^{-2})
$SW\uparrow$	outgoing shortwave radiation (W m^{-2})
$SW\downarrow$	incoming shortwave radiation (W m^{-2})
T	air temperature (K, °C)

t	time (s)
w	vertical wind speed (m s^{-1})
Δx	element thickness (m)
z_0	roughness length (m)
z_d	zeroplane displacement (m)
z_h	mean building height (m)
α	Albedo
β	Bowen ratio (Q_H/Q_E)
Θ	potential temperature (K)
λ_p	plan area index.
λ_V	vegetation fraction
ρC	volumetric heat capacity (J m^{-3} K^{-1}))
ρ	air density (kg m^{-3})
ρ_v	water vapor density (kg m^{-3})
MO	Monin-Obukhov
NDVI	Normalized Difference Vegetation Index
OHM	Objective Hysteresis Model
RES	Residual approach for determining ΔQS
SEB	Surface energy balance
SVF	Sky view factor
TEB	Town Energy Balance Model
TMS	Thermal Mass Scheme
UCZ	Urban Climate Zones
UEB	Urban energy balance
UHI	Urban heat island
UTZ	Urban Terrain Zones

0.3.1. Notation for Part 2

Symbol	Meaning
ΔA	Net moisture advection
ΔS	Net change in storage
ΔS_m	Net change in soil moisture
ΔS_n	Net change in snowpack storage
ΔW	Net change in ground water storage
E	Evapotranspiration
E_T	Transpiration
E_V	Evaporation
F	Anthropogenic water release
F_C	Anthropogenic water release due to combustion
F_H	Anthropogenic water release due to urban heating systems
F_M	Anthropogenic water release due to cooling and air conditioning
F_W	Consumption of bottled/imported water
I	Piped water supply
I_G	Water use for irrigation.
I_R	Water supplied by water management techniques
I_S	Supplied water from source
I_U	Water usage
P	Precipitation
r	Runoff
r_F	Surface infiltration
r_L	Surface runoff (e.g. overland flow and roof runoff).
r_O	Snow melt runoff
r_S	Storm water runoff (through storm drainage system)
r_W	Wastewater drainage
S_A	Anthropogenic water storage
S_W	Surface water storage
W	Ground water storage
W_I	Water supply pipe leakage to/from groundwater
W_S	Sewer and storm drain pipe leakage to/from groundwater.

0.4 Document References

0.4.1. References for Part 1: Energy

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0.5 Project Overview

Urban metabolism considers a city as a system and distinguishes between energy and material flows. "Metabolic" studies are usually top-down approaches that assess the inputs and outputs of food, water, energy, etc. from a city, or that compare the metabolic process of several cities. In contrast, bottom-up approaches are based on quantitative estimates of urban metabolism components at local scale, considering the urban metabolism as the 3D exchange and transformation of energy and matter between a city and its environment. Recent advances in bio-physical sciences have led to new methods to estimate energy, water, carbon and pollutants fluxes. However, there is poor communication of new knowledge to end-users, such as planners, architects and engineers.

BRIDGE aims at illustrating the advantages of considering environmental issues in urban planning. BRIDGE will not perform a complete life cycle analysis or whole system urban metabolism, but rather focuses on specific metabolism components (energy, water, carbon, pollutants). BRIDGE's main goal is to develop a Decision Support System (DSS) which has the potential to propose modifications on the metabolism of urban systems towards sustainability.

BRIDGE is a joint effort of 14 Organizations from 11 EU countries. Helsinki, Athens, London, Firenze and Gliwice have been selected as case study cities. The project uses a "Community of Practice" approach, which means that local stakeholders and scientists of the BRIDGE meet on a regular basis to learn from each other. The end-users are therefore involved in the project from the beginning. The energy and water fluxes are measured and modelled at local scale. The fluxes of carbon and pollutants are modelled and their spatio-temporal distributions are estimated. These fluxes are simulated in a 3D context and also dynamically by using state-of-the-art numerical models, which normally simulate the complexity of the urban dynamical process exploiting the power and capabilities of modern computer platforms. The output of the above models lead to indicators which define the state of the urban environment. The end-users decide on the objectives that correspond to their needs and determine objectives' relative importance. Once the objectives have been determined, a set of associated criteria are developed to link the objectives with the indicators. BRIDGE integrate key environmental and socio-economic considerations into urban planning through Strategic Environmental Assessment. The BRIDGE DSS evaluates how planning alternatives can modify the physical flows of the above urban metabolism components. A Multi-criteria Decision Making approach has been adopted in BRIDGE DSS. To cope with the complexity of urban metabolism issues, the objectives measure the intensity of the interactions among the different elements in the system and its environment. The objectives are related to the fluxes of energy, water, carbon and pollutants in the case studies. The evaluation of the performance of each alternative is done in accordance with the developed scales for each criterion to measure the performance of individual alternatives.

Several studies have addressed urban metabolism issues, but few have integrated the development of numerical tools and methodologies for the analysis of fluxes between a city and its environment with its validation and application in terms of future development alternatives, based on environmental and socio-economic indicators for baseline and extreme situations. The innovation of BRIDGE lies in the development of a DSS integrating the bio-physical observations with socio-economic issues. It allows end-users to evaluate several urban planning alternatives based on their initial identification of planning objectives. In this way, sustainable planning strategies will be proposed based on quantitative assessments of energy, water, carbon and pollutants fluxes.

Part I: Energy in the urban system

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1.1 Introduction

Today, energy consumption is highest in urban areas. While exact estimates vary, cities are estimated to use up to three-quarters of the world's commercial energy. While natural resources of fossil fuels are declining worldwide, their use still dominates and renewable energy technologies currently contribute only a small share. To develop sustainable cities and sustainable urban planning, several challenges in energy use have to be faced by urban planners and developers - increasing efficiency of use, a general reduction in demand, and a maximization of the share of renewable energy should be the main topics. To address these challenges, information on the distribution and flows of energy in typical urban systems have to be known. The energy consumption of a city is strongly dependent on the prevailing local climate and its urban built-up structure. Together these define the microclimate within the street canyons, on the roads, in the buildings, and at any other place in an urban area.

Energy is almost everywhere; it exists in different forms and can be transported, stored and transformed. Electricity, heat, radiation, fuels, materials, evaporation, emissions – distinguishing between what is energy and what, for example, material is difficult. An attempt often followed by ecologists to integrate the energy that is contained in material, energy and information flows into a consistent measurement scheme is done by the concept of "embodied energy". The embodied energy, called emergy, can provide a single measurement that accounts for all the mentioned flows within an urban material metabolism and between this metabolism and its surroundings (e.g. Odum *et al*. 1996, Huang 2005, Zhang *et al*. 2009).

Decker (2000) distinguished flows of energy and matter through urban metabolism by their type of input: active inputs through human work and passive inputs, e.g. through solar radiation. An interesting approach based on information and entropy analysis is outlined by Balocco & Grazzini (2006). It follows the second law of thermodynamics for analyzing urban systems and weights factor/actor interactions to derive a basis for decision makers where interventions would have the highest energy sustainability.

The concept of urban metabolism (Wolman 1965, Boyden *et al*. 1981, Girardet 1992, Newman 1999) regards a city as a system and usually distinguishes between energy and material flows as its components. Within the framework of BRIDGE, the focus lies on the energy, water and carbon, and pollutants. Energy enters passes and leaves the system in several ways and in several physical states and forms. Fuels (like coal, oil and natural gas), electricity (generated from different sources), radiation and heat are the main categories. But even construction materials, food, water and waste may be considered as stored energy. The definition of what is regarded as energy depends on the point of view: urban planners, city administrations, economists and statisticians often have a different perspective to natural scientists, like meteorologists or physicians. The first have to deal primarily with fluxes of "usable energy", the optimization of use and questions such as how energy consumption can be influenced by administrative means, for example, guidelines for insulation of new houses, old-building renovation, traffic reduction etc. The latter are more interested in understanding how energy in the form of radiation and heat influences the urban climate, how it is transported and stored (e.g. in urban built-up structures).

Within each of those approaches the other one is normally only marginally included. If there is an attempt to include different perspectives, for a single planning approach or within a certain study for example, usually one part is generally summarized to a single factor in the system. Energy flow charts by urban planners usually omit radiation as a heating source, and the anthropogenic share in the emitted radiation as well as in the atmospheric heat fluxes are part of those fluxes that are summarized as losses of the system. From the micrometeorological point of view though, those losses are one input factor to the urban energy balance, summarized under the term anthropogenic heat flux (e.g. Oke 1987, Roberts *et al*. 2006) (discussed further below).

One opportunity of BRIDGE is the reduction of the gap between urban planners and environmental scientists and the better integration of natural sciences into planning. The focus is on those processes that form the interface between the two different views - mainly size, pattern, variability of the anthropogenic heat flux and its dependence on the urban structure.

In this paper we give an overview on the urban energy flows primarily from a meteorological point of view. The topics are mainly restricted to the atmospheric fluxes and their dependence on the urban structure. Recent research efforts and state of the art methods and models are reviewed.

1.2 The "meteorological" view – Urban Energy Balance

The energy balance of an urban system (hereafter referred to as urban energy balance - UEB) can be determined in a micrometeorological sense by considering the energy flows in and out of a control volume. For such a control volume reaching from ground to a certain height above the buildings, the energy balance equation can be defined (e.g. Oke 1987; Offerle *et al*. 2005):

$$Q^* + Q_F = Q_H + Q_E + \Delta Q_S + \Delta Q_A + S$$

where Q^* is the net all wave radiation, Q_F is the anthropogenic heat flux, Q_H is the turbulent sensible heat flux, Q_E is the turbulent latent heat flux, ΔQ_S as the net storage change within the control volume, ΔQ_A is the net advected flux ($Q_{A\,in}$ - $Q_{A\,out}$), and S is all other sources and sinks. All terms are usually expressed as energy flux density per horizontal or vertical area (typically W m^{-2}, also MJ m^{-2} d^{-1} for temporal sums). For comparisons between sites it is common to non-dimensionalize fluxes (Table 1.2); expressing individual terms as a percentage of the net radiation, which besides the comparatively small and difficult to determine Q_F is the main input term into the system. In the following sections each of the UEB terms is examined separately.

Studies or models may refer to the surface energy balance (SEB) (e.g. Grimmond & Oke 2002, Lemonsu et al. 2004) instead of the UEB. The two terms are generally interchangeable, but the reader has to pay attention to the way the exchange surface has been defined in each case. The SEB may sometimes be regarded as distinct from the UEB in the fact that it can consider the natural or built surface of the earth (e.g. soil, roads, roofs or walls) as the border or plane where exchange processes take part and not the top of a building-air box volume. Storage change in this case would be the flux into or out of the ground, whereas the advection term would fall out of the equation. This interpretation of the SEB is often applied for microscale considerations over more or less homogeneous surfaces.

1.3 Scales

The urban atmosphere usually is divided into vertical layers as illustrated in Figure 1.1. The lower atmosphere that is influenced by the urban structure is called the Urban Boundary Layer (UBL). From the ground up to roughly the average height of roughness elements like buildings or trees (z_H) is the urban canopy layer (UCL). It is produced by micro-scale processes which characterize their immediate surroundings. The UCL is part of the roughness sublayer (RS) which is dependent on the height and density of roughness elements and extends to $z^* = a\, z_H$, where a ranges between 2 and 5 (Raupach *et al*. 1991). Above this is the inertial sublayer (IS) where under ideal conditions the Monin-Obukhov Similarity Theory may be expected to apply (although this is normally not the case in urban areas - see later discussion). The upper part of the UBL, which is to a large extent determined by meso-scale advective processes, is referred to as the outer urban boundary layer (Rotach *et al*. 2005).

A key issue of importance for urban investigations is the definition of the appropriate scale of a study area. A proposal of how the UCL elements can be classified according to scale considerations is given in Table 1.1 (Oke 2005). It should be noted that for investigations of specific surface materials, for example the ability of concrete to store or release heat, the scale of study might be smaller than the size of a building.

Today, the energy fluxes in urban areas are being investigated by three main fields of natural science: micrometeorological site studies, remote sensing measurements and modeling approaches. Each of those fields has its advantages but also its restrictions or methodological imperfections. In combination they yield complimentary insights; each is considered here.

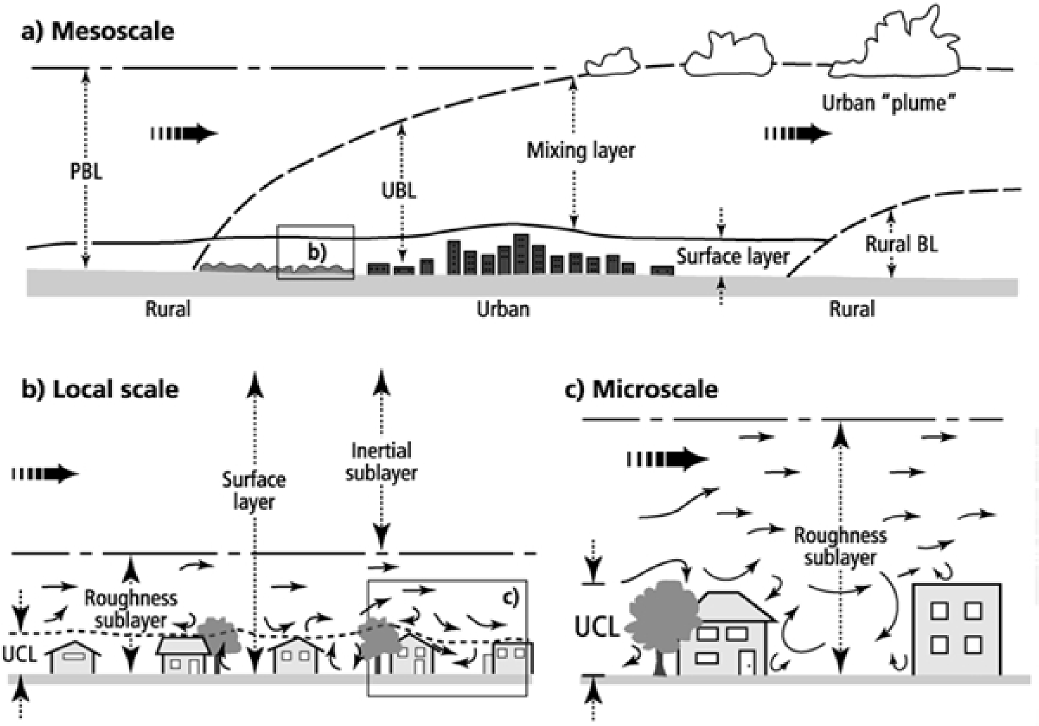

Figure 1.1: Meteorological scales used in urban areas (from Piringer *et al.* 2002).

Table 1.1: Elements of the urban canopy layer (UCL) and their scales (Oke 2005).

UCL units	Built features	Typical horizontal length scales
1. Building	Building	10 m × 10 m
2. Canyon	Street, canyon	30 m × 40 m
3. Block	Block, factory	0.5 km × 0.5 km
4. Land-use class or UTZ or UCZ[1]	City centre, residential, or industrial zone	5 km × 5 km
5. City	Urban area	25 km × 25 km
6. Urban region	City plus its environs	100 km × 100 km

[1] Urban Terrain Zones (Ellefsen 1990/91) or Urban Climate Zones (Oke 2006)

1.4 Measurement and Modelling of Energy Balance Fluxes

1.4.1 Net all wave radiation (*Q**)

Net radiation, *Q**, is the net balance between the incoming (↓) and outgoing (↑) short (*SW*) and long wave (*LW*) radiation fluxes:

$$Q^* = SW\downarrow - SW\uparrow + LW\downarrow - LW\uparrow$$

Measurement can be made using pyranometers for the shortwave fluxes and pyrgeometers for the longwave fluxes, or by using net radiometers. High quality instruments have instrumental errors less than 4 %, or 20 W m^{-2}, whichever is larger. The height of the radiation measurement determines the size of the source area and thus the representativeness of the measurement (Oke 2006).

In a typical urban atmosphere radiative fluxes are altered by pollutants. Whereas *SW*↓ will be reduced, the returning *LW*↓ is greater. In typical mid-latitude cities, these changes are normally opposed by a lower short wave albedo due to darker surface materials (whereas in low-latitude cities walls and roofs are generally brighter) and a higher surface temperature at night, which augments the long wave emission (Oke 1987). The net effect on urban/rural radiation differences therefore remains small (Oke 1987, Rotach *et al*. 2005). Christen & Vogt (2004) found the net radiation for Basel, Switzerland to be nearly equivalent on average over urban and rural surfaces, with urban albedo values in the order of 10 % (0.1).

When modeling the radiative fluxes of the surface energy balance, the focus lies on how the outgoing radiative fluxes are determined as the incoming radiative fluxes typically will be prescribed. The surface properties considered (i.e. the degree of detail) also are important. A major difference between models relates to the number of reflections assumed. Table 1.3 gives an overview of methods used by UEB models to model outgoing shortwave and long wave radiation. The range is from single bulk values (one albedo value for an area, e.g. LUMPS) to multiple reflections in canyons (e.g. BEP02/05, SUMM) (Grimmond *et al*. 2007). For model abbreviations and detailed descriptions of the models see Table 1.17. Montavez *et al*. (2000) found that neglecting multiple reflections in their Monte Carlo model can lead to errors in the outgoing radiation of up to 10 %.

Table 1.2: Flux ratios for daytime conditions at different sites. For explanations on the flux terms please refer to the respective chapters following below. β is the Bowen ratio Q_H/Q_E. All quantities are non-dimensional. Values given in parentheses for $\Delta Q_S/Q^*$ show results after daytime turbulent fluxes are adjusted so that storage is zero over the period (DOY 229–246 2002, Offerle *et al*. (2006)). Data from Cleugh and Oke (1986), Oke *et al*. (1992), Oke (1999), Grimmond & Oke (1999), Christen (2005), Offerle *et al*. (2006), Masson et al. (2008), Lemonsu et al. (2008), Pearlmutter *et al*. (2009), Christen *et al*. (2009).

City	Site (year)	Land-use	Q_H/Q^*	Q_E/Q^*	$\Delta Q_S/Q^*$	β
Vancouver (CDN)	Vs 1983	Residential	0.44			
Mexico City (MEX)	TA 1985	Dense urban, mixed	0.34			
Vancouver (CDN)	Vs 1989	Residential	0.54	0.27	0.19	2.0
Tucson (US)	T 1990	Residential	0.52	0.25	0.23	2.1
Sacramento (US)	S 1991	Residential	0.41	0.33	0.26	1.3
Vancouver (CDN)	VI 1992	Industrial	0.42	0.10	0.48	4.4
Vancouver (CDN)	Vs 1992	Residential	0.62	0.22	0.17	2.9
Mexico City (MEX)	SM 1993	Central city	0.38	0.04	0.58	9.9
Los Angeles (US)	A 1993	Residential	0.39	0.31	0.30	1.2
Los Angeles (US)	Sg 1994	Residential	0.49	0.22	0.29	2.2
Los Angeles (US)	A 1994	Residential	0.43	0.26	0.31	1.4
Miami (US)	Mi 1995	Residential	0.42	0.27	0.30	1.6
Chicago (US)	C 1995	Residential	0.46	0.37	0.17	1.2
Basel (CH)	U1 2002	Residential	0.54	0.22	0.34	2.55
	U2 2002	Residential/commercial	0.55	0.24	0.37	2.47
Łódź (PL)	CBD 2002	Commercial, institutional, residential	0.44	0.23	0.32 (0.29)	1.83
	IND 2002	Industrial	0.36	0.21	0.41 (0.31)	1.61
	RES 2002	Residential	0.30	0.37	0.32 (0.23)	0.80
	RUR 2002	Rural, Airport	0.22	0.54	0.24 (0.16)	0.41
Toulouse (F)	Tl 2004-05	Old European city center	0.86 to 1.78	0.26 to 0.49		3.28 to 3.69
Montréal (CDN)	MUSE 2005	Dense residential (snow)	0.32	0.08		7.82
		(no snow)	0.44	0.04		10.39
Vancouver (CDN)	Sunset 2008	Residential	0.59	0.17		3.43
	Oakridge 2008	Residential	0.45	0.25		1.86
	Westham I. 2008	Rural, reference	0.38	0.33		1.13
Negev (ISR)	Artificial	1-storey (A_W/A_H = 0)[1]	0.56	0.00	0.44	132.45
	Open-Air	1-storey (A_W/A_H = 0.1)	0.45	0.21	0.35	2.15
	Scaled Urban	1-storey (A_W/A_H = 0.2)	0.38	0.37	0.25	1.02
	Surface	2-storey (A_W/A_H = 0)	0.52	0.01	0.47	89.91
	(OASUS)	2-storey (A_W/A_H = 0.1)	0.52	0.12	0.36	4.38
		2-storey (A_W/A_H = 0.2)	0.44	0.29	0.27	1.52

[1] A_W/A_H is the proportion of water surface area to total horizontal area.

Several studies (see Table 1.4) have focused on detailed investigation and modeling of the radiative fluxes at the micro scale (e.g. Montavez et al. 2000, Kusaka et al. 2001, Offerle *et al*. 2003, Blankenstein & Kuttler 2004, Kastendeuch & Najjar 2009). The prediction of surface temperatures within whole urban areas and for specific street canyons in particular is a main objective of such radiation models. Here, a main determining factor besides surface properties is the urban geometry. A good parameter to describe this geometry and to estimate radiative fluxes for street canyons (e.g. the longwave downward radiation (Nunez *et al*. 2000, Blankenstein & Kuttler 2004)) is the Sky View Factor (SVF). Different estimation alternatives have been examined and compared by Grimmond *et al*. (2001), Chapman & Thornes (2004) and Rigo & Parlow (2007). The ratios of

building height to street width and length has been evaluated by Herbert & Herbert (2002) using a numerical model.

Even if the canyon geometry is known to have an impact on all radiative flux terms, models represent it with different accuracy. The bandwidth ranges from simple schemes that use bulk surface parameterizations without distinguishing between different surface types (e.g. the net all-wave radiation parameterization (NARP) (Offerle *et al*. 2003), to schemes that include distinct surfaces (e.g. roofs, walls, roads) each with their own radiative characteristics and reflections of radiation within the canyon (Town Energy Balance (TEB) scheme (Masson 2000)), to high resolution numerical calculations of single canyons (Kastendeuch & Najjar 2009).

A comprehensive study with a high resolution 3D-numeric simulation and validation of the algorithms and results of the radiative fluxes in an urban street canyon was recently conducted by Kastendeuch & Najjar (2009) for the city of Strasbourg, France. Heat transfer algorithms were taken from scientific literature. The simulation of LW↑ was do ne by an iterative solution of the energy balance equation. The authors conclude that this procedure seems to correctly predict the surface temperatures and the shortwave multiple reflections within the canyon play an important role. If coupled with a CFD (Computational Fluid Dynamics) model, a highly accurate prediction of the UCL air temperature is expected to be possible. Kastendeuch & Najjar (2009) note the importance of the improvement of the non-radiative heat transfers methods in order to apply the model to any kind of surface.

Using remote sensing methods, spatial upward radiation patterns can be measured for larger areas or whole cities. By analyzing TERRA/ASTER data, Chrysoulakis (2003) estimated Q^* for the center of the metropolitan city of Athens within about +/-44.5 W m-2. Urban land surface temperatures were estimated for Los Angeles and Paris by Dousset & Gourmelon (2003) joining NOAA-AVHRR thermal images and SPOT-HRV derived landcover classification with a GIS and in-situ measurement data.

In addition to generally known limitations of remote sensing techniques, such as restricted field of view (FOV), angle and shadowing effects or impacting vegetation, additional obstacles have to be considered if radiative fluxes are being determined from air- or space-borne measurements. First, the upward surface radiation is not hemispheric but directionally variable due to the urban structure. As a consequence the surface temperature estimated by remote sensing varies with sensor viewing geometry and is subject to an anisotropy error that increases with increasing canyon aspect ratio (Sugawara & Takamura 2006). Voogt & Oke (1998) measured variations of the surface temperature of more than 9 °C with different sensor viewing geometries for the downtown area of Vancouver, Canada, while Kobayashi and Takamura (1994) estimated errors for a nadir-view sensor with a narrow FOV (a sensor that does not see walls) to be 0.5-1.5 K. The error related to long wave anisotropy may be a key component to the successful closure of the energy balance in urban systems (Sugawara & Takamura 2006).

1.4.2 Turbulent sensible heat flux (Q_H)

The turbulent sensible heat flux – as all vertical fluxes in the surface boundary layer - can be estimated by two different basic approaches. The first approach is the flux-gradient method where the flux is derived from average vertical profiles of the corresponding scalar, e.g. air temperature in the case of sensible heat flux. The second is known as the eddy covariance method which tries to sense the properties of eddies passing through a measurement level at a high frequency. The flux of sensible heat can be written for the gradient method as follows (Oke 1987):

$$Q_H = -\rho c_p K_H \frac{\partial \bar{\theta}}{\partial z}$$

where ρ is the air density (kg m^{-3}) c_p is the specific heat capacity of air (J kg^{-1} K^{-1}), K_H is the eddy conductivity (m^2 s^{-1}) and $\partial \bar{\theta}/\partial z$ the vertical gradient of the potential temperature. Based on the Monin-Obukhov similarity theory, stability corrections have to be applied for diabatic conditions. Using high frequency instruments (e.g. ultrasonic anemometer-thermometers) the eddy covariance method can be applied. The respective eddy covariance form of Q_H can be summarized as:

$$Q_H = \rho c_p \overline{w'T'}$$

where $\overline{w'T'}$ is the average of the product of the turbulent fluctuations of air temperature *T* and the vertical wind speed *w*. The primes denote the deviations from the mean and the overbar the average.

Table 1.3: Methods used by UEB models to model outgoing shortwave and long wave radiation. For model descriptions and references see Table 1.17. (Grimmond *et al*. 2009a)

Model	**Number of reflections**	**Albedo**	***LW*↑**
BEP02	multiple	canyon	Result of multiple reflections in the canyon, from walls and canyon floors at each grid level
BEP05	multiple	canyon	Result of multiple reflections in the canyon
CAT	one	by facet	By facet, with one emission and one reflection among facets
CLMU	multiple	by facet	By facet with multiple reflections and one emission
ENVImet	one	by facet	From energy balance of all facets
GCTTC	multiple	by facet	Prescribed
HIRLAM-U	one	bulk/town	Fast broadband schemes for solar and thermal radiation (Savijärvi 1990) emissivity function and Stefan-Boltzmann law, STRACO scheme for clouds, water vapour (Sass 2002)
LUMPS	one	bulk	Prata (1996)
MM5u	one	bulk/town	Stefan-Boltzmann law, plus parameterization schemes for the clouds and water vapour (e.g. Stephens 1978; Garand 1983)
MOSES1T	one	bulk	Prescribed bulk emissivity
MOSES2T	one	canyon, roof	Prescribed emissivity values for canyon and roof
MOUSES	multiple	bulk/effective	Effective emissivity by multiple reflections
MUCM	two	by facet	
MUKLIMO			From energy balance of all facets
NSLUCM	one		One reflection
SM2U	infinite	bulk/effective	Effective emissivity
SUEB	one	bulk/town	By emissivity
SUMM	multiple		Result of multiple reflections
SUNBEEM	multiple	By facet	By facet with multiple reflection
TEB, TEB07	infinite	canyon, roof	Result of two reflections in the canyon
TUF2D, TUF3D	multiple (min 2)	patches /facet	Multiple reflection (minimum 2) between patches (and emission by patches initially)
UCLM	two		Emitted longwave radiation computed for each facet (no reflections) using energy budget.
VUCM	three		One reflection

For inhomogeneous urban areas, the eddy covariance method is more suitable, as flux-gradient relationships and the corresponding stability corrections normally fail in the roughness sublayer (i.e. the diffusivities for heat and water vapour differ) (Roth & Oke 1995, Piringer 2002, Christen 2005). To carry out measurements higher up, in the inertial sublayer, is difficult in urban areas due to a lack of higher towers and because of fetch considerations. An overview over typical values for Q_H in urban areas is presented in Table 1.6.

For model calculations Q_H can be estimated by different methods. A basic empirical scheme to estimate hourly Q_H from routine weather data during daytime was introduced by Holtslag & van Ulden (1983) and is still used in many energy balance studies (e.g. Batchvarova & Gryning 1991, or Hanna & Chang 1992, Grimmond & Oke 2002). Input weather parameters are air temperature, cloud cover and wind speed, and Q_H und Q_E are partitioned (Bowen ratio, $\beta = Q_H/Q_E$) using the Penman-Monteith approach (Monteith 1981). De Bruin & Holtslag (1982) compared the Penman-Monteith method with a modified approach of the Priestley-Taylor method (Priestley & Taylor 1972) and found that under the prevailing conditions, both were able to appropriately describe the hourly latent and sensible heat fluxes.

The LUMPS model (Grimmond & Oke 2002) calculates the turbulent heat fluxes based on the available energy. Sensible and latent fluxes are partitioned using the approach of de Bruin & Holtslag (1982) and Holtslag & van Ulden (1983) with a new scheme to define the α and β parameters for urban environments (Table 1.5). Field

observations in seven North American cities (from the multicity urban hydrometeorological database, MUHD) have been used to evaluate LUMPS and illustrate its broad utility (Grimmond & Oke 2002).

A comparison of LUMPS and an Aerodynamic Resistance Method (ARM) was carried out by Xu et al (2008). Here, surface parameters were derived from 6 m spatial resolution airborne hyperspectral imagery for Shanghai, China. Q_H was found to be largest for rooftops (~350 W m^{-2}). Spatially averaged values for both approaches were found to be rather similar which is encouraging for further applications as the two methods have different data requirements and operate in distinct ways. If it is taken into account that shadow effects and anthropogenic heat fluxes affect the use of the LUMPS model, air- or spaceborne thermal remote sensing data combined with externally-sourced land cover information are a practicable solution for heat flux determination in urban areas (Xu *et al*. 2008). If remotely sensed surface temperatures are used, the directional variation of the radiation has to be considered (Sugawara & Takamura 2006).

Differentiations between surface and air temperatures show similar spatial and temporal patterns but not an exact correspondence. Dependence on microscale site characteristics is much stronger for surface temperatures, especially under calm, clear, nocturnal conditions with minimal turbulent interactions.

Table 1.4: Typical values for Q^* ($W\ m^{-2}$) from literature.

Authors	Site	Period	Q^* Daily	Q^* daytime	Remarks
Oke 1987	Sheffield	1952	56		
	Montréal	1961	52		Summer: 57; Winter: 153
	Los Angeles	1965-70	108		
	Fairbanks	1965-70	18		
	Manhatten	1967	93		Summer: 40; Winter: 198
	Berlin	1967	57		
	Budapest	1970	46		Summer:100; Winter: -8
	Vancouver	1970	57		Summer: 107; Winter: 6
	Hong Kong	1971	~110		
	Singapore	1972	~110		
Grimmond & Oke 1999	Tucson	Jun 1990	144.7	189.2	
	Sacramento	Aug 1991	112.5	146.5	
	Vancouver suburban	Jul/Sep 1992	102.8	140.4	
	Vancouver Industrial	Aug 1992	132.1	161.5	
	Los Angeles, Arcadia	Jul/Aug 1993	159.0	178.7	
	Mexico City	Dec 1993	39.1	100.8	
	Los Angeles, Arcadia	Jul 1994	180.3	202.4	
	Los Angeles, San Gabriel	Jul 1994	144.1	169.6	
	Chicago	Jun/Aug 1995	172.3	195.1	
	Miami	May/Jun 1995	159.0	184.8	
Oke *et al.* 1999	Mexico City (cloudless)	Dec 1993	42.1	103.9	Avg. peak: 400; Nighttime: after sunset –120, near sunrise –90
Piringer 2002	MUHD sites		Average max.: < 400-650		
Offerle *et al.* 2003	Chicago (clear sky)	1992/93		158	nighttime: -31
	Los Angeles (clear sky)	1993/94		255	nighttime: -48
	Łódź (clear sky)	2001		137	nighttime: -38
Christen 2005	Basel Urban	2001/02	57.4		
	Basel Rural	2001/02	57.4		
	Basel Urban1	Jun/Jul 2002	145.8	482	nighttime: -65
	Basel Urban2	Jun/Jul 2002	150.5	481	nighttime: -62
	Basel Urban3	Jun/Jul 2002	74.1	322	nighttime: -81
	Basel Suburban	Jun/Jul 2002	142.4	453	nighttime: -56
Offerle *et al.* 2006	Łódź , CBD	Year 2002	99.6	292.1	
	Łódź , Industrial	Aug 2002	82.9	264.0	
	Łódź , Residential	Aug 2002	107.7	304.9	
	Łódź , Rural	Aug 2002	133.2	290.2	
Barzyk & Frederick 2008	Chicago University	Aug 2005	93.3		
	Chicago City Hall	Aug 2005	49.2		
Masson *et al.*, 2008	Toulouse	Spring (Mar-May 2004)	81		
		Summer (Jun-Aug 2004)	123		
		Fall (Sep-Nov 2004)	33		
		Winter (Dec 2004-Feb 2005)	-1		
Lemonsu *et al.* 2008	Montréal, Dense residential	Mar 2005 (snow)		337	
		Apr 2005 (no snow)		389	
Christen *et al.* 2009	Vancouver Sunset	Aug 2008	131.3		
	Vancouver Oakridge	Aug 2008	119.4		
	Vancouver Westham Island (rural)	Aug 2008	150.6		
Pearlmutter *et al.* 2009	1-storey (A_W/A_H = 0)[1]	Summer	111.6	132.4	Artificial Open-Air Scaled Urban Surface (OASUS), arid Negev region, southern Israel.
	1-storey (A_W/A_H = 0.1)		115.0	139.1	
	1-storey (A_W/A_H = 0.2)		121.8	146.8	
	2-storey (A_W/A_H = 0)		125.5	151.7	
	2-storey (A_W/A_H = 0.1)		129.6	156.3	
	2-storey (A_W/A_H = 0.2)		144.7	164.1	

[1] A_W/A_H is the proportion of water surface area to total horizontal area.

Table 1.5: Methods used by UEB models to calculate sensible heat flux (Q_H). MO = Monin-Obukhov. For model descriptions and references see Table 1.17 (Grimmond *et al*. 2007).

Model	Turbulent sensible heat flux methods
BEP02, BEP05, SUNBEEM	For walls based on Clarke (1985).
CAT	Resistance between canyon surfaces and air based on Hagishima & Tanimoto (2003); at canyon top depends on stability, using empirical parameterization.
CLMU	Resistance network accounting for differences between surfaces.
ENVImet	From turbulence model (wall function) and surface energy balance.
GCTTC	Calculated for each surface based on the attenuated radiation by the CTTC factor.
LUMPS	De Bruin & Holtslag (1982) modified Penman-Monteith for urban areas (Grimmond & Oke 2002).
HIRLAM-U, MM5u	Parametric formulation based on the specific heat capacity for moist air, the density of the atmosphere, the surface friction velocity and the surface temperature scale.
MOSES2T, MOSES1T	Standard resistance, based upon MO similarity theory.
MOUSES	Resistance network based on Harman *et al*. (2004).
MUCM	MO or Jürges (1924).
MUKLIMO	From surface energy balances at the soil, walls and roofs using MO laws.
NSLUCM	MO based on Louis (1979) and Jürge's (1924) formulation, and calculated from each surface.
SM2U	MO resistance (Guilloteau, 1998, Zilitinkevich, 1995).
SUEB	MO similarity Louis (1979) modified by Mascart *et al*. (1995).
SUMM	Resistance (top-down method, Kanda *et al*. 2005).
TEB, TEB07	Resistance.
TUF2D, TUF3D	Resistances based on flat-plate heat transfer coefficients (vertical patches) and based on MO similarity (horizontal patches).
UCLM	Exchange based on canyon air and surface temperature difference, wind speed and prescribed heat transfer coefficient.
VUCM	Parametric formulation.

1.4.3 Turbulent latent heat flux (Q_E)

In urban areas, the fraction of the turbulent heat fluxes that is Q_E depends primarily on the availability of water and thus on the presence of vegetated areas (transpiration) or wet surfaces (evaporation). Similar to the sensible heat flux this flux can be defined as (Oke 1987):

$$Q_E = -L_v K_V \frac{\partial \overline{\rho_v}}{\partial z}$$

with L_v the latent heat of vaporization (J kg^{-1}), K_V the eddy diffusivity for water vapor (m^2 s^{-1}) and $\partial \overline{\rho_v}/\partial z$ the vertical gradient of the water vapor density (kg m^{-3} m^{-1}). The respective eddy covariance form of Q_E is written as:

$$Q_E = L_v \overline{w'\rho'_v}$$

Q_E can be measured directly using fast responding instruments (e.g. an ultrasonic anemometer-thermometer coupled with an open path infrared gas analyzer) or calculated using the Penman-Monteith or the Priestley-Taylor method (see Q_H section and section II). In models, Q_E again is calculated in different ways. Some models ignore the flux completely (assuming a dry urban area), others have wet built surfaces but no vegetation and some include vegetation either as a separate tile or as integrated (see also section 1.5). Methods for calculating Q_E typically involve some form of resistance scheme or prescribe a value based on areal extent of vegetation (e.g. GCTTC) (see Grimmond *et al*. 2007 for details).

If irrigation is low, and the partitioning of the turbulent fluxes at a rural reference is known, vegetation fraction is a suitable input parameter for the estimation of the partitioning in simple models (Christen 2005). Remote sensing (e.g. aerial photos, NDVI) or maps are a good basis to easily derive vegetation fraction for an urban area as a model input.

Table 1.6: Typical values for Q_H (W m^{-2}) from literature. For additional values see flux ratios in Table 1.2.

Authors	**Site**	**Q_H Daily**	**Q_H daytime**
Asaeda 1996	Tokyo 1990/91, Summer	Over asphalt: max. 350 Over concrete: max. 200	
Grimmond & Oke 1999	Tucson 1990	89.5	98.3
	Sacramento 1991	57.6	60.8
	Vancouver Industrial 1992	69.2	68.3
	Vancouver suburban 1992	84.3	86.8
	Los Angeles, Arcadia 1993	69.3	69.0
	Mexico City 1993	41.8	38.7
	Los Angeles, Arcadia 1994	86.8	86.0
	Los Angeles, San Gabriel 1994	85.2	82.5
	Chicago 1995	87.5	89.6
	Miami 1995	77.4	78.2
Oke *et al*. 1999	Mexico City (cloudless) 1993	42.25	39.4
Piringer 2002	MUHD sites	Average daily max.: 120-310	
Offerle *et al*. 2006	Łódź, CBD 2002	55.8	127.5
	Łódź, Industrial 2002	39.3	94.5
	Łódź, Residential 2002	36.6	90.9
	Łódź, Rural 2002	27.6	64.7
Barzyk & Frederick 2008	Chicago University 2005	64.7	
	Chicago City Hall 2005	22.1	
	Vancouver Oakridge 2005	64.8	
	Westham Island (rural) 2005	57.9	
Masson *et al.*, 2008	Toulouse, Spring 2004	83	
	Toulouse, Summer 2004	105	
	Toulouse, Fall 2004	59	
	Toulouse, Winter 2004-2005	61	
Lemonsu *et al*. 2008	Montréal, Dense residential 2005		107 (snow) 170 (no snow)
Christen *et al*. 2009	Vancouver Sunset 2008	94.9	
Pearlmutter *et al*. 2009	1-storey (A_W/A_H = 0)[1]	74.4	74.0
	1-storey (A_W/A_H = 0.1)	60.1	62.2
	1-storey (A_W/A_H = 0.2)	52.4	55.6
	2-storey (A_W/A_H = 0)	82.6	79.2
	2-storey (A_W/A_H = 0.1)	82.1	80.8
	2-storey (A_W/A_H = 0.2)	73.0	72.1

[1] A_W/A_H is the proportion of water surface area to total horizontal area.

The quantification of evapotranspiration for an extended urban area is again difficult as sources of moisture are extremely heterogeneous. An experiment in an artificial scaled urban surface with evaporation pans of varying size (with A_W/A_H as the proportion of water surface area to total horizontal area) embedded in building arrays of varying height, was carried out in an arid Negev region of southern Israel (Pearlmutter *et al*. 2005, 2007, 2009). The relation between the available water and latent heat removal was found to be almost linear (see Table 1.7) and accompanied by an approximately equal decrease in storage and turbulent sensible heat flux (Pearlmutter *et al*. 2009). Such experiments may help considerably in understanding and quantifying urban turbulent heat flux patterns within complex urban areas and may present a valuable tool for testing and verifying the different calculation methods used by models (see Table 1.8).

This term is discussed further in section II, when evapotranspiration (its water equivalent) is considered.

1.4.4 Net storage change within a control volume (ΔQ_S)

The net heat storage flux ΔQ_S consists of the uptake or release of energy by the ground, buildings and vegetation (surface energy balance, micro-scale) and includes also the changes of latent and sensible heat content in the air of the considered control volume (local-/meso-scale). The latter changes are often neglected as they are small compared to the heat storage changes in urban materials.

The rate of change of heat storage ΔQ_S within an urban control volume can be expressed as the sum of storage fluxes for single surface elements i (Offerle *et al*. 2005):

$$\Delta Q_S = \sum_i \frac{\Delta T_i}{\Delta t} (\rho C)_i \Delta x_i \lambda_{pi}$$

where $\Delta T/\Delta t$ is the rate of temperature change, ρC is the volumetric heat capacity, Δx is the element thickness and λ_p is the plan area index. $\Delta x \lambda_p$ is the total element volume over the plan area for each element i (Offerle et al. 2005).

Table 1.7: Typical values for Q_E (W m^{-2}) from literature. For additional values see flux ratios in Table 1.17.

Authors	**Site**	**Q_E Daily**	**Q_E daytime**
Grimmond & Oke 1999	Tucson 1990	56.7	47.2
	Sacramento 1991	50.7	48.1
	Vancouver Industrial 1992	17.1	15.5
	Vancouver suburban 1992	31.0	30.3
	Los Angeles, Arcadia 1993	57.1	55.4
	Mexico City 1993	3.6	3.9
	Los Angeles, Arcadia 1994	54.4	53.4
	Los Angeles, San Gabriel 1994	40.0	38.0
	Chicago 1995	78.7	72.2
	Miami 1995	53.0	50.7
Oke *et al*. 1999	Mexico City (cloudless) 1993	4.1	4.5
Piringer 2002	MUHD sites	Average daily max.: 10-235	
Offerle *et al*. 2006	Łódź ,CBD 2002	38.8	68.4
	Łódź, Industrial 2002	31.6	56.6
	Łódź, Residential 2002	58.5	112.5
	Łódź, Rural 2002	89.4	156.8
Masson *et al.*, 2008	Toulouse, Spring 2004	24	
	Toulouse, Summer 2004	32	
	Toulouse, Fall 2004	16	
	Toulouse, Winter 2004-2005	14	
Lemonsu *et al.* 2008	Montréal, Dense residential 2005		26 (snow) 16 (no snow)
Christen *et al*. 2009	Vancouver Sunset 2008	33.6	
	Vancouver Oakridge 2008	44.0	
	Vancouver Westham Island (rural) 2008	56.7	
Pearlmutter *et al*. 2009	1-storey (A_W/A_H = 0)[1]	0.6	-2.0
	1-storey (A_W/A_H = 0.1)	28.8	29.1
	1-storey (A_W/A_H = 0.2)	54.6	60.2
	2-storey (A_W/A_H = 0)	0.9	-1.3
	2-storey (A_W/A_H = 0.1)	18.4	22.9
	2-storey (A_W/A_H = 0.2)	47.5	52.5

[1] A_W/A_H is the proportion of water surface area to total horizontal area.

As cities are not expected to cool down or heat up during a year, the yearly total of ΔQ_S has to be zero by definition (Christen 2005, Offerle et al. 2005). This is helpful in calculating annual surface energy balances and in assigning annual residuals to other terms as, for example, the anthropogenic heat flux. Even, it should tend to very low values over a day period (Offerle et al. 2005, Pigeon et al. 2007b) which allows to estimate the anthropogenic heat flux for entire daily periods.

ΔQ_S is a spatially and temporally variable term of the energy balance, depending on differences in surface type and radiant loading. Even on cloud-free days variations in the convective fluxes induce hourly variability (Grimmond & Oke 2002). It is of particular relevance in the urban energy balance as it can account for more than half of the daytime net radiation at highly urbanized sites (Roberts *et al*. 2006). Measurements of ΔQ_S at non-urban sites are normally obtained with ground heat flux plates. Obviously, direct measurements in urban areas are practically unattainable due to the complexity of urban structures and materials. It therefore has to be determined by indirect methods or models.

A commonly used indirect method is to consider the storage flux term as the residual (RES) of the energy balance (e.g.; Roth & Oke 1994, Grimmond & Oke 1995, 1999, Christen & Vogt 2004, Spronken-Smith *et al*. 2006):

$$\Delta Q_S = Q^* - Q_H - Q_E$$

Q_A, Q_F and S are here considered as negligible. As for most fluxes that are not directly measureable, there is a lack of standard for the determination of urban heat storage and quite a range of determination methods exist. A widely used parameterization approach, on which the Objective Hysteresis Model (OHM) (Oke & Cleugh 1987, Grimmond *et al*. 1991) is based, uses relations between the net-all wave radiation Q^* and the storage heat flux ΔQ_S for typical surface materials. Based on an earlier approach of Oke (1981), Camuffo & Bernardi (1982) formulated a hysteresis relation between Q^* and Q_G (or ΔQ_S) that, summed over each component urban surface type (i), gives the equation:

$$\Delta Q_S = \sum_{i=1}^{n}\left[a_{1i}Q^* + a_{2i}\left(\frac{\partial Q^*}{\partial t}\right) + a_{3i}\right]$$

Values for the three regression coefficients (a_1, a_2, a_3) have been published for several types of urban surfaces (Grimmond *et al*. 1991, Pearlmutter *et al*. 2004, Christen 2005, Roberts *et al*. 2006) and are partially summarized in Meyn & Oke (2009).

Table 1.8: Methods used by different UEB models to calculate latent heat flux (Q_E) and soil moisture. MO = Monin-Obukhov. For model descriptions and references see Table 1.17. (from Grimmond *et al*. 2009)

Model	Latent heat flux method	Soil moisture method
BEP02	Not included.	Not included.
CAT	Resistances based on the Penman-Monteith equation (Grimmond & Oke 2002).	Not included.
CLMU	Resistance network accounting for differences between surfaces.	Layers
GCTTC	Evapotranspiration per 1 m^2 of vegetated coverage estimated empirically Shashua-Bar & Hoffman (2002) method.	Not included.
HIRLAM-U	Bulk method over each subgrid-scale surface type and tile method for flux aggregation.	Force-restore, 2 soil layers + forest canopy, ISBA scheme (Noilhan & Planton 1989).
LUMPS	DeBruin & Holtslag (1984) modified Penman-Monteith.	Not included.
ENVImet	Soil hydrological model, at Surface Halstead parameter calculated, Vegetation Photosynthesis/Transpiration model.	Prognostic 1-d multilayer model.
NSLUCM	Using land-surface model for latent heat fluxes from natural surfaces, and bulk/slab model for evaporation from anthropogenic surfaces.	Prognostic multi-layer soil model for natural surfaces and one-layer slab model for athropogenic surface.
MUCM	Conductance based on Schulze *et al*. (1994).	Not included.
MM5u	Parametric formulation based on the heat of vaporization, the available moisture, the molecular diffusivity, the depth of the molecular layer and the specific humidity at the surface and the lowest model level.	Five-Layer Soil Model (Dudhia 1996).
MOSES1T MOSES2T	Standard resistance based upon MO similarity theory.	Prognostic 1-d multilayer model.
MUKLIMO	Resistance law within the canopy, MO from there to the atmosphere.	Prognostic 1-d multilayer model.
SM2U	Resistance (Noilhan & Planton 1989).	Force-restore, 2 layers + reservoir.
MOUSES	Resistance	Prognostic 1D multilayer model.
TEB	Resistance	Bucket
TEB07		Bucket
TUF2D, TUF3D	Not included.	Not included.
BEP05	Penman-Monteith formulation (Monteith 1981).	Force-restore.
SUEB	Resistances based on Best (1998).	Not included.
SUMM	Resistance	Not included.
SUNBEEM	Penman-Monteith formulation (Monteith 1981).	No explicit soil moisture. Thermal properties for vegetated canyon floor may reflect soil moisture.
UCLM	Not included.	Not included.
VUCM	Parametric formulation.	Layers

A comparison of different methods to estimateΔ Q_S has been undertaken by Roberts *et al*. (2006) for the study site of Marseille, France, for eight days in summer. They compared the RES approach with the OHM parameterization scheme, a local-scale numerical model (Town Energy Balance model, TEB), and a bulk heat transfer method - the Thermal Mass Scheme (TMS). The old tall buildings of Marseille's city center, with thick

walls, represent a large thermal mass and favor heat storage. Normally the uncertainties of the RES method depend on the fraction of Q_E in the heat fluxes. Due to the Mediterranean warm and dry climate at Marseilles Q_E was almost negligible, which reduced the uncertainties in ΔQ_S. The results for the three models compared to the RES approach all showed a similar diurnal course, but heat storage release at night was generally underestimated. RES estimates derived from high-quality observational studies are often considered of being closest to the correct value (Roberts *et al*. 2006).

A remote sensing way to calculate and model ΔQ_S for the city of Basel was followed by Rigo & Parlow (2007). They tried to combined data from several satellites with in-situ measurements using three different approaches. An empirical regression function derived from NDVI-values, the OHM approach, and a calculation of the complete aspect ratio (CAR) from a high resolution urban surface model. The results from all three approaches showed similar performance and corresponded quite well with the measured values. Rigo & Parlow (2007) mention the OHM to be the most promising and favor it for further use because of its good transferability if similar urban parameters are given.

Grimmond & Oke (2002) found ΔQ_S to be most important at downtown and light-industrial sites that are comparatively dry and built-over, representing at least 50% of daytime Q^*. For residential sites, values commonly are 20-30 % of Q^*. The nighttime upward-directed flux is initially larger than the net radiation but is reduced after one or two hours to ±5 % the net radiation loss.

Urban built-up surfaces are mainly distinguishable into three types: ground/roads, walls and roofs. In mid-latitude urban areas, roofs in general have a comparatively low heat capacity and thus represent a minor contribution to ΔQ_S. Heat storage characteristics for 6 different roof assemblies in Vancouver, Canada, were investigated across a range of wind and moisture conditions by Meyn & Oke (2009). A conclusion from this study was that the contribution of roofs to total heat storage may be relatively minor compared to walls and ground, including roads (Meyn & Oke 2009). This is supported by the results of the TMS model for Marseilles where the roofs represent the largest built-up volume (35 %) and are highly exposed to insolation but have a relatively low heat capacity and a high albedo. Thus, heat storage in this case is less important than the loss of heat by convection due to exposed ventilation (Roberts *et al*. 2006).

Because of their low albedo (0.08 for Marseilles after Roberts *et al*. 2006) and their intermediate heat capacity, roads are energetically more significant than roofs or walls. The overall daily contribution to the bulk storage heat flux depends mainly on their orientation, the aspect ratio and the resulting diurnal insolation pattern. Xu et al (2008) found peak values for ΔQ_S to be around 420 W m^{-2} for Shanghai.

An overview of different methods to estimate and calculate ΔQ_S is given in Table 1.9. Meaningful error analysis of the different methods is almost impossible as each is, after Roberts *et al*. (2006), open to one or more sources of error:

- instrumental or methodological imperfection (RES, OHM, TEB, TMS (see Table 1.9 and Table 1.17 for method abbreviations and descriptions))
- the physical impossibility of conducting adequate sampling (OHM, TMS)
- the inadequacy and oversimplification inherent in parameterizations and the non transferability of their coefficients (OHM, TEB)
- the limits of numerical methods (TEB)
- evaluations that are also unable to appeal to a standard (OHM, TEB, TMS).

Table 1.9: An overview of different general methods to determine ΔQ_S and an open list of authors that used them.

General method	Authors
Residual calculation (RES) from SEB	Roberts *et al*. 2006, Kato & Yamaguchi 2007
Parameterization scheme (e.g. OHM)	Grimmond *et al*. 1991, Arnfield & Grimmond 1998, Grimmond & Oke 1999, Taha 1999, Erell & Williamson 2006, Roberts *et al*. 2006, Spronken-Smith *et al.* 2006, Sugawara & Takamura 2006, Pearlmutter *et al*. 2007, Rigo & Parlow 2007, Coutts *et al*. 2008, Xu *et al*. 2008, Meyn & Oke 2009
Multi-Layer thermal diffusion for different surface types	Wilkes 1989, Masson 2000, Kusaka *et al*. 2001, Masson *et al*. 2002, Offerle *et al*. 2003, Lemonsu *et al*. 2004, Roberts *et al*. 2006, Hamdi & Masson 2008, Pigeon *et al*. 2008, Meyn & Oke 2009
Bulk heat transfer method (e.g. TMS)	Roberts *et al*. 2006
Calculation from urban surface model (e.g. CAR (Complete Aspect Ratio))	Rigo & Parlow 2007
Empirical regression function (e.g. NDVI (Normalized Difference Vegetation Index))	Rigo & Parlow 2007
Monte Carlo Method	Montavez & Jimenez 2000

Table 1.10: Typical values for ΔQ_S (W m^{-2}) from literature by year of publication. For additional values see flux ratios in Table 1.2.

Authors	Site	ΔQ_S Daily	ΔQ_S daytime	ΔQ_S Nighttime
Grimmond & Oke 1999	Tucson 1990	232.9	43.8	
	Sacramento 1991	4.1	37.7	
	Vancouver Industrial 1992	45.7	77.7	
	Vancouver suburban 1992	244.1	23.3	
	Los Angeles, Arcadia 1993	32.6	54.3	
	Mexico City 1993	237.7	58.2	
	Los Angeles, Arcadia 1994	39.1	63.1	
	Los Angeles, San Gabriel 1994	18.9	49.2	
	Chicago 1995	6.1	33.3	
	Miami 1995	28.8	56.0	
Oke *et al*. 1999	Mexico City 1993	-4.2	60.2	
Piringer 2002	MUHD sites Av. daily max.:	150 - 280		
Offerle *et al*. 2005	Łódź 2001-02		Monthly averages, range: -8 to +6 Yearly average: -0.01 to -0.04	
Offerle *et al*. 2006	Łódź, CBD 2002	5.0	96.1	
	Łódź, Industrial 2002	12.0	113.0	
	Łódź, Residential 2002	12.6	101.5	
	Łódź, Rural 2002	16.3	68.6	
Roberts *et al*. 2006	Marseilles (RES) 2001	-9	67	-99
	Marseilles (OHM) 2001	27	84	-42
	Marseilles (TEB) 2001	10	97	-94
	Marseilles (TMS) 2001	-2	41	-54
Pearlmutter *et al*. 2009	1-storey (A_W/A_H = 0)[1]	39.1	57.9	
	1-storey (A_W/A_H = 0.1)	25.9	48.0	
	1-storey (A_W/A_H = 0.2)	9.1	36.6	
	2-storey (A_W/A_H = 0)	44.1	71.6	
	2-storey (A_W/A_H = 0.1)	24.7	56.9	
	2-storey (A_W/A_H = 0.2)	19.1	44.4	

[1] A_W/A_H is the proportion of water surface area to total horizontal area.

1.4.5 Net advected flux (ΔQ_A)

Mathematically, advection is the transport of a scalar in a fluid, which can be described as a vector field for this purpose. The scalar may be any kind of substance or conserved property (like heat) in a transporting fluid (air). Storage change in a control volume due to advection can be written as result of the flow in and out of the volume as:

$$\Delta Q_A = Q_{in} - Q_{out}$$

Advection is a term that has been neglected for a long time in measurement studies and models as its influence was considered to be very small and often the theoretical assumption of horizontal homogeneity was adopted. Obviously this is not the case for local scale meteorological investigations in urban areas. Still, many urban heat balance studies neglect the advection term even if its impact on the energy balance might be considerable for certain study sites (Spronken-Smith *et al*. 2006, Pigeon *et al*. 2007a).

Table 1.11: Methods used by different UEB models to calculate storage heat flux (ΔQ_S). For model descriptions and references see Table 1.17 (from Grimmond et al. 2007).

Model	Storage Heat Flux methods
BEP02	Sum of storage heat flux of roofs, walls, and road for each energy budget solved
CAT, HIRLAM-U, MM5u, LUMPS	OHM scheme (Grimmond *et al.* 1991)
ENVImet	Soil 1D model, fully resolved, walls/building system no storage term
GCTTC	Residual from radiation after allowing for sensible and latent heat.
MUCM	Finite difference
MUKLIMO	Walls and roofs have a heat capacity
NSLUCM	Multi-layer thermal diffusion within wall/roof/road
SM2U	Budget + Conduction + Force restore
SUMM	Multi-layer thermal diffusion within walls/roof/road
SUEB	As Q_G in urban slab (solution of multi layer thermal diffusion equation)
SUNBEEM	1D finite difference solution of heat conduction equation for each facet
BEP05, TEB, TEB07, CLMU, BEP05, TUF2D, TUF3D, MOSES2T, MOSES1T, VUCM, MOUSES	Diffusion (in different resolutions, e.g. within wall/roof/road)
UCLM	Substrate heat exchange & changes substrate temperatures accordingly.

Pigeon et al. (2007a) found sea-breeze circulations to determine the flow during afternoon periods for Marseilles, France. Horizontal advection of heat was up to 100 W m^{-2}. The surface sensible heat flux from the SEB is then different from the sensible heat flux at the top of the control volume considered in the UEB (see former discussion). Then the horizontal advection of heat (100 W m^{-2}) must be added to the measured sensible heat flux (at 28 m above the roofs) to retrieve the surface heat flux. On the contrary, the moisture advection converted to equivalent latent heat flux suggested an overestimation by the measurements of the surface latent heat flux of 50 W m^{-2}. Thus horizontal advection can be high for coastal cities like Marseilles during summertime. Its influence on the energy balance is lower in magnitude for cities with other topographical surroundings, e.g. hills and valleys with cold drainage flows. Spronken-Smith et al. (2006), for example, concluded thatΔ Q_A was not greater than 30-40 W m^{-2} in magnitude for stable wintertime conditions in Christchurch, New Zealand.

If numerical simulation is used to evaluate horizontal advection, meso-scale atmospheric models can be used to derive size and direction of the advection itself and, coupled with local urban energy balance models, to estimate their influence on the local scale (e.g. Meso-NH coupled with TEB-ISBA in Pigeon, 2007a).

For future measurements, as well for improved modeling approaches, it might be of importance that advective fluxes are considered, especially for cities that are influenced by noticeable time-constant flow patterns, e.g. diurnal wind systems like sea-breeze circulations or drainage flows.

In a number of field experiments in forests the influence of advection on the mass and energy balance of control volumes was investigated applying an advection completed mass balance (Aubinet *et al*, 2009). Results up to now show, that there is a large uncertainty in quantifying horizontal and vertical advection fluxes. Both terms are large, are coupled and seem not to cancel each other. It is not known yet how relevant this is for the urban environment.

Table 1.12: Typical values for ΔQ_A from literature.

Authors	Site	ΔQ_A	
Spronken-Smith *et al.* 2006	Christchurch, NZ		max. 30-40 Wm^{-2}
Pigeon *et al.* 2007a	Marseilles, France	Heat: Moisture:	max. 100 Wm^{-2} max. - 50 Wm^{-2}

1.4.6 Anthropogenic heat flux (Q_F)

The anthropogenic heat flux Q_F consists mainly of combustion exhausts by stationary and mobile sources (Roberts *et al.* 2006). Thus, its contribution to the UEB is highest in wintertime when the energy input from human sources is comparatively large (primarily due to domestic heating). But even in summertime it may become significant for cities with a high air conditioning usage. Q_F is difficult to determine because of its strongly varying pattern and because it cannot be measured directly. It is therefor not surprising that many different approaches to estimate this term of the UEB can be found in literature (summarized in Table 1.14). The following paragraphs give an overview.

Inventories

A common approach is to estimate Q_F based on inventories of existing socio-economic data. Most inventory-based estimations derive Q_F from energy use data. For example, early estimations based on the per capita energy use for several cities are summarized in Oke (1990) (see Table 1.14). Estimations based on surrogates for energy use data such as dwelling density, numbers in employment and service and industrial floor area have been made for London for the 1970ties by Harrison *et al*. (1984).

A more complex inventory method is applied in Grimmond (1992), where Q_F is approximated on an hourly base utilizing different parameters for the three main contributors: traffic, stationary sources and metabolic release. The fraction of traffic as a function of type and amount of gasoline used, the number of vehicles, their fuel efficiency and the traveled distance. For stationary sources consumer-scale long term data was dissolved to short term fluctuations using large scale hourly variations of electricity and gas consumption. And the metabolic release was calculated using metabolic rates from Oke (1978) for people and 25 per cent of that for animals (Bach 1970).

Sailor & Lu (2004) estimated diurnal profiles of heat released by buildings, transportation and metabolism based on a population density formulation for six large US cities. During summer they found heating from vehicles to be the dominant term for every city (47 – 62%) while during winter heating fuels accounted for 51 – 57% in the cold climate cities of Philadelphia, Salt Lake City and Chicago. Human metabolic heat release was the least important with 2-3%. The same method was used to estimate Q_F in an UHI experiment for the less dense built-up city of Rome by Bonacquisti *et al*. (2006). They found Q_F to be around 20 W m^{-2} on a summer day and 100 W m^{-2} on a winter day but it was concluded that this does not play an important role in the UHI development.

A more population-independent approach based on energy consumption by building categories and per unit floor area was applied by Moriwaki *et al*. (2008) in their detailed analysis of the temporal and spatial variations in anthropogenic heat and water vapor emissions in Tokyo. They found the daytime anthropogenic sensible heat flux to origin mainly from offices while at nighttime it was mainly from residential housing.

For Marseilles, Grimmond *et al*. (2004) estimated the daily course and peaks of Q_F based on observed CO_2 fluxes as an independent measure of human (largely traffic) activity. Results were found to vary between 15 W m^{-2} (nighttime) and 50 W m^{-2} (daytime), with peaks around 75 W m^{-2} during rush-hours.

Residual of the energy budget

A second approach to estimate Q_F is to calculate it as the residual of the UEB. This method is a more physical one, but all errors of the other terms are contained in the residual. For example energy balance closure has to be assumed although it is well known, that the turbulent heat fluxes are typically underestimated when the eddy-covariance method is used. Another problem is, that the storage term is also calculated as the residual of the UEB. If daily or yearly totals of the energy balance equation are considered then ΔQ_S can be neglected and Q_F calculated as the residual term is a reasonable method (Christen & Vogt 2004).

For the city of Łódź, Poland, Q_F has been determined using monthly energy consumption data from the mid-1980's by Kłysik (1996) and as residual of the SEB by Offerle *et al*. (2005). Both found mean Q_F to be larger in winter than in summer and the latter stated that it shows an inverse relationship with air temperature. But the size of the estimated fluxes was of a remarkable difference (see Table 1.14), which could be due to the fact that the studies were twenty years apart. Recently Pigeon *et al.* (2007b) conducted a study on Q_F with a direct comparison of the two methods and showed that they can lead to similar results. During winter Q_F is estimated by both methods to be around 70 W m^{-2} while during summer the residual method yields 15 W m^{-2} compared to 30 W m^{-2} from the inventory method. Pigeon *et al.* (2007b) attribute this to the fact that Q_H and Q_E are known to be underestimated by the eddy-covariance technique, especially as the turbulent heat fluxes are higher during the summer period.

Models

Implementation of Q_F into UEB models can be done in different ways, including the mentioned ones, as Table 1.13 shows. The term most typically is prescribed although some components may be calculated, e.g. fixed or

mobile sources (Grimmond *et al*. 2007). For example, a method to estimate Q_F within a TEB model study for Toulouse applied recently by Pigeon (2008) is to analyze traffic counts and an inventory of energy consumption for buildings. This inventory was built on real consumption of electricity and gas (90% of energy use in Toulouse buildings). For the other sources of energy (domestic fuel, wood and others), mean annual values were taken. Mean findings of this study were that for winter and fall, releases from buildings (space heating and other domestic sources) dominate Q_F (see also Pigeon 2007b). Validations of Q_F showed that the estimations done with the inventory resulted in a good reproduction of the mean value (difference of 1 W m^{-2}) for the autumn period, whereas lowest values were under-predicted and highest over-predicted. Heat release associated with space heating was suspected to be probably a strong contributor to slightly positive nighttime Q_H. In winter, space heating releases were probably too high, but the TEB model was generally able to reproduce the order of magnitude (80 W m^{-2}) and processes associated with this term.

Impacts on urban climate

The spatial and temporal patterns of Q_F, its impact on the urban climate and its implications for urban planners are still prone to large uncertainties. Masson (2006) considers large Japanese cities as good targets for meso-scale studies as they have a high density of population, high buildings and large energy consumption, and consequently a large contribution of Q_F to the UEB. Ichinose et al. (1999) for example concluded that anthropogenic fluxes can explain the areas of maximum UHI effect in Tokyo and that reducing energy consumption by 50% for hot water supply and 100% for space cooling could lead to -0.5°C air temperature reduction. As a general outlook, Grimmond et al. (2007) argue that Q_F may become especially significant at key times of the day and night and specifically at transition times. Therefor, the detailed role of Q_F within the UEB is yet to be fully investigated and this is a future key research field for urban energy studies.

Table 1.13: Methods used by different UEB models to determine anthropogenic heat flux Q_F. For model descriptions and references see Table 1.17. (from Grimmond et al. 2007).

Model	Anthropogenic heat flux methods
BEP02, BEP05, SUNBEEM	Partially accounted for by imposing a fixed temp at the building interior.
CAT	Prescribed, adjusted for diurnal variations.
CLMU	Prescribed traffic fluxes, parameterized waste heat fluxes from heating/ air conditioning.
ENVImet	From heat transfer equations through walls.
HIRLAM-U	Calculated (offline) as a temporal & spatial function of available parameters by 4 methods (emission, night light, land-use, population) (Baklanov *et al*. 2005).
GCTTC	Prescribed per vehicle (for vehicles only).
MM5u, NSLUCM	Calculated (offline) as a temporal & spatial function of the anthropogenic emissions.
MOSES2T, MOSES1T, MOUSES, SUEB	Not modelled itself but possible to be included for calculation of turbulent fluxes.
MUCM	Modelled by Kondo & Kikegawa (2003) offline.
MUKLIMO	Heat fluxes through the walls and roofs computed with fixed temp at the buildings interior.
TEB, TEB07 TUF2D, TUF3D	Prescribed traffic and industrial fluxes, domestic heating computed from internal building temperature force-restore equation and conduction through the wall and the roof.
UCLM	Not directly included. Heat can be added to building interior.
VUCM, SM2U	Prescribed bulk value.

1.4.7 Other sources and sinks (*S*)

Based on current research knowledge on the urban energy balance, it seems accurate to consider a term (*S*) in the UEB, under which all yet neglected or unknown processes can be summarized. These processes individually will be comparatively small but one possible contributor to this term may be rainwater, which absorbs heat from the surface but is channelled out of the system via sewers (Offerle *et al*. 2005). Further investigation on such processes itself and the necessity of including them into the UEB are clearly needed.

Table 1.14: Typical values for Q_F (W m^{-2}) and methods used by year of publication.

Authors	**Site**	Q_F	**Method**
Myrup 1969	Typical U.S. city	Year: 91	Estimates. After Harrison *et al*. (1984).
Kalma 1974	Sydney	Winter: < 1 – 25 (7-9 a.m.)	
Torrance & Shum 1976	Typical U.S. city	Year: Central Area: 84 Year: Summer: 75; Winter: 92; Diurnal: Summer: 28 – 100; Winter: 58 – 120	
Llewellyn & Washington 1977	Major U.S. city	Summer: 21-42	
Koenig 1979	Washington D.C. Chicago Houston	Year: 46 Year: 22 Year: 12	
Smith 1976	8 British towns	Year: 20 – 70	Subjectivly assessed industrial factor. After Harrison *et al*. (1984).
Reichenbächer 1978	Berlin	Winter: 10.1 Summer: 5.3	SO_2 as an indicator. After Harrison *et al*. (1984).
Summers 1963	Montréal	Winter: 117 – 157	Inventories. After Harrison *et al*. (1984).
Bornstein 1968	Manhattan	Winter: 199	
Bach 1970	Cincinatti	Year: 26	
Tapper *et al*. 1981	Christchurch	Winter: 3.9	
Harrison *et al*. 1984	London 1971-76	Year: max. 187 Hourly: < 1 to 333 per grid space	1 km^2 grid. Surrogates for energy use: dwelling density, no. in employment, service/industrial floor area.
Oke 1987	Sheffield 1952 Montréal 1961 Los Angeles 1965-70 Fairbanks 1965-70 Berlin 1967 Manhattan 1967 Budapest 1970 Vancouver 1970 Hong Kong 1971 Singapore 1972	Year: 19 Year: 99 Summer: 57; Winter: 153 Year: 21 Year: 19 Year: 21 Year: 117 Summer: 40; Winter: 198 Year: 43 Summer: 32; Winter: 51 Year: 19 Summer: 15; Winter: 23 Year: 4 Year: 3	Based on estimates of per capita energy use within city from all sources.
Grimmond 1992	Vancouver, Canada 1987	Daily (Jan-Jun): 7 – 9 Less during daytime	Approximated using traffic and energy consumption data and metabolic heat release parameters.
Kłysik 1996	Łódź, Poland 1984-86	annual average: 40 January: 71 July: 18	Based on monthly energy consumption data
Ichinose 1999	Tokyo, Japan 1989	Winter daytime: 400 Winter morning peak: 1590	Data from energy statistics and land use data set/building storeys.
Grimmond *et al*. 2004	Marseilles, France 2001	nighttime: 15 daytime: 50 peaks: 75	Based on CO_2 fluxes as measure of human activity.
Sailor & Lu 2004	Chicago, San Francisco, Philadelphia 2002 Atlanta, Salt Lake City 2002	Winter: 70 – 75 Summer: Peaks: 30-60 Peaks: < 15	Calculated heat released from buildings, transportation and metabolism based on population density and other data sources. Higher magnitudes for urban cores postulated.
Christen & Vogt 2004	Basel, Switzerland 2002	annual average: 10 - 20	Residual of annual SEB
Offerle *et al*. 2005	Łódź, Poland 2001-02	Summer: -3 Winter: 32 Mean Q_F shows inverse relationship with air temperature.	Residual of SEB. The physically unrealistic negative value may suggest underestimation of turbulent fluxes. Error resulting from neglecting changes in heat storage should be within ±10 W m^{-2}.
Pigeon et al. 2007b	Toulouse 2004-05	Spring 26 Summer 15 Fall 42 winter 76	Residual of SEB and inventory of energy consumption.
Moriwaki 2008	Tokyo, Japan	Daily: 5-35 Summer Max.: 284 Anthropogenic sensible heat only	Estimated from energy consumption statistics for 250x308m grid cells.

1.5 Urban energy balance studies and models

The field of urban climate research has provided a large number of studies and papers during the last decades. A comprehensive overview over studies on the urban energy balance has been given in the preceding chapters on the terms of the UEB. Table 1.15 summarizes these studies.

Most reviews on UEB studies and models generally provide an overview only of a part of the field, a specific methodology, or are restricted to only a range of scales. A more general review of the progress in urban climatology over the preceding two decades is given by Arnfield (2003). He addresses exchange processes for energy at scales up to neighborhoods and explores the literature on urban temperature as two main topics amongst others. During the last years, significant advances have been made in the correct simulation of the physics of urban climate processes associated with higher computational capacity and an increased ability to couple the schemes with atmospheric models (Masson 2006).

Local scale models can be classified by different attributes and characteristics. In a review of the current status of urban SEB models, Masson (2006) classified newer SEB models into five categories: (1) models statistically fitted to observations; (2) modified vegetation schemes with drag terms in the canopy; (3) without drag terms; (4) new urban canopy schemes that present both horizontal and vertical surfaces, with a drag approach; (5) without a drag approach. He also noted that empirical models, as they reflect observations, are useful to interpret results of more complex schemes. And that they might be sufficient for study cases where good observational understanding exists. Multi-layer models become necessary when objectives are more specific and e.g. feedbacks between different processes get included.

Other classification schemes are presented in Grimmond *et al*. (2009a) and summarized in Table 1.16. A spatial allocation pattern for example is the orientation of the canyon, whether it is considered shaded at appropriate times of the day (e.g. CLMU, CAT, SUNBEEM) or one default wall is considered as representative for the whole model (e.g. TEB). The resolution of distinct built facets is a further criterion. A general differentiation is between roofs, walls and roads. Detailed models also calculate spatial variability along facets (e.g. TUF3D).

Most of the characteristics that are being used for classifications have been discussed in the preceeding sections. One characteristic that has been mentioned in the Q_E section is the role of the urban vegetation. It is, if at all, incorporated in two different ways: as a separate tile, not interacting with the surface beyond the first layer of the meso-scale model (e.g. TEB, MOSES) or integrated into the urban area and affected by, as well as affecting, the built environment (e.g. CLMU, SUNBEEM, LUMPS). According to Grimmond *et al*. (2007) the modeling of the vegetation itself is done using separate, well tested vegetation models and resistance schemes developed for urban areas (e.g. Grimmond and Oke 1991, Arnfield 2000).

Depending on the complexity of a model, processes may not always explicitly be simulated but parameterized in a model. Instead of simulating for example the roughness sublayer, the urban surface may be described by modified surface properties. Such properties were derived by many urban SEB studies, e.g. for a “European” city (Basel, Switzerland) by Christen (2005). Here, typical properties for urban surfaces were found to be (with z_h the mean building height): zeroplane displacement $z_d = 0.8\ z_h - 0.9\ z_h$, roughness length $z_0 = 0.1\ z_h$, albedo α = 10%, vegetation fraction $\lambda_V = 0.2$, and sky view factor at ground level = 0.4. For suburban surfaces the values were: $z_d = 0.6\ z_h$, $z_0 = 0.3\ z_h$, α = 13%, $\lambda_V = 0.5$, and sky view factor = 0.6.

Classifications and analyses of potential strengths and weaknesses of models help to find an appropriate model for a specific study case. However, comparisons of only the characteristics of models give no evidence on how accurate they actually predict the terms of the UEB. Only a direct comparison of models being fed with the same input parameters will give that evidence. An extensive urban surface energy balance model comparison to evaluate different schemes is now being carried out (Grimmond *et al*. 2007). The adopted methodology follows that used in PILPS, the Project for Intercomparison of Land-Surface Parameterization Schemes (Henderson-Sellers *et al*. 1993). The four key questions being investigated in this systematic evaluation are (Grimmond *et al*. 2007):

- What are the main physical processes that need to be resolved to simulate realistically urban energy balance exchanges and do the models produce realistic simulations?
- How complex does a model need to be in order to produce a realistic simulation of urban fluxes and temperatures? Balancing needs, costs and performance.

- Which input parameter information is required by an urban model to perform realistically? What is the minimum number of parameters required, what are these parameters and how accurately do they need to be known?
- What are the main research priorities for future observational campaigns within urban areas? What are the essential parameters to be measured in observational studies, what are the appropriate techniques?

The main goal of the project is to gain insight into the strengths and weaknesses of different classes of urban models. An overview of the models being investigated is given by Table 1.17. All models in the comparison consider a single point within an urban area (i.e. are one-dimensional) and run offline. First results of the model comparison study have been published (Grimmond *et al.* 2009a,b,c). While the net radiation seemed to be reasonably well calculated by all models, heat storage results showed a broad variability. The sensible heat flux was generally overestimated by the models and the latent heat flux results had the lowest correlation coefficient (Grimmond *et al*. 2009b,c).

The default way to verify the output of a model is a comparison with measured variables. As micrometeorological site measurements represent only point data, their representativeness within the strong heterogeneous urban canopy is limited. Variable outputs of meso-scale models are representatives of spatial averages and therefore not directly comparable. Martilli (2007) pronounces CFD models as a possible way to bridge this gap. CFD results for small areas can be validated against point measurements, be spatially averaged and then compared to meso-scale model results.

A future challenge in modeling the UEB is that most simulations have yet been done for mid-latitude cities with their typical climate variability. Improvements in the models have to be made to better adapt them to other climates or special climatic situations, for example, the presence of snow that has impact on the partitioning of the turbulent and the radiative fluxes (Martilli 2007). The TEB scheme of Masson (2000) accounts for snow but has not yet been tested extensively for this aspect, except for Montréal, Canada (Belair 2006, Mailhot *et al*. 2006, Leroyer *et al*. 2009, Lemonsu *et al*. 2009). In his concluding remarks Arnfield (2003) states that '*Energy balance evaluations should continuingly be extended to non-temperate climates and additionally to central city sites; For* Q_F *and F* (Anthropogenic Water Release due to Combustion, see Part II, Section 2.6) *additional empirical observations and methods of estimation are required, e.g. how traffic counts and population data can be embedded into climatic research; Methods for the linkage between scales (small and meso) have to be investigated; Validation of models has to be improved and collaboration between modelers and field climatologists has to be intensified*.'

Table 1.15: Urban energy balance studies by year (if no year given, publication yearis listed). The'x' indicates if term was measured. For details and values of the terms please see respective sections of this paper.

Authors	Site	Year	Q^*	Q_H	Q_E	Q_S	Q_A	Q_F
Oke 1987	Sheffield	1952	x					x
Oke 1987	Montréal	1961	x					x
Summers 1963	Montréal	1963						x
Oke 1987	Los Angeles	1965-70	x					x
Oke 1987	Fairbanks	1965-70	x					x
Oke 1987	Manhatten	1967	x					x
Oke 1987	Berlin	1967	x					x
Bornstein 1968	Manhattan	1968						x
Myrup 1969	Typical U.S. city	1969						x
Bach 1970	Cincinatti	1970						x
Oke 1987	Budapest	1970	x					x
Oke 1987	Vancouver	1970	x					x
Oke 1987	Hong Kong	1971	x					x
Harrison et al. 1984	London	1971-76						x
Oke 1987	Singapore	1972	x					x
Kalma 1974	Sydney	1974						x
Torrance & Shum 1976	Typcial U.S. city	1976						x
Smith 1976	8 British towns	1976						x
Llewellyn & Washington 1977	Major U.S. city	1977						x
Reichenbächer 1978	Berlin	1978						x
Koenig 1979	Washington D.C.	1979						x
Koenig 1979	Chicago	1979						x
Koenig 1979	Houston	1979						x
Tapper et al. 1981	Christchurch	1981						x
Cleugh & Oke 1986	Vancouver	1983	x	x				
Klysik 1996	Lodz	1984-86						x
Oke et al. 1999	Mexico City (MEX)	1985	x	x				
Grimmond 1992	Vancouver	1987	x	x	x	x		x
Oke et al. 1999	Vancouver	1989	x	x	x	x		
Ichinose 1999	Tokyo	1989						x
Grimmond & Oke 1999	Tucson	1990	x	x	x	x		
Asaeda 1996	Tokyo	1990/91	x	x	x	x		
Grimmond & Oke 1999	Sacramento	1991	x	x	x	x		
Grimmond & Oke 1999	Vancouver	1992	x	x	x	x		
Offerle et al. 2003	Chicago	1992/93	x					
Grimmond & Oke 1999	Los Angeles	1993	x	x	x	x		
Grimmond & Oke 1999, Oke et al. 1999	Mexico City	1993	x	x	x	x		
Offerle et al. 2003	Los Angeles	1993/94	x					
Grimmond & Oke 1999	Los Angeles	1994	x	x	x	x		
Grimmond & Oke 1999	Chicago	1995	x	x	x	x		
Grimmond & Oke 1999	Miami	1995	x	x	x	x		
Spronken-Smith et al. 2006	Christchurch (NZ)	1995	x	x	x	x	x	
Offerle et al. 2005	Lodz	2001	x			x		x
Roberts et al. 2006	Marseilles	2001	x	x	x	x		
Pigeon 2007a	Marseilles	2001	x	x	x	x	x	
Christen 2005	Basel	2001/02	x	x	x	x		
Offerle et al. 2006	Lodz	2002	x	x	x	x		x
Christen & Vogt 2004, Christen 2005	Basel	2002	x	x	x	x	x	x
Sailor & Lu 2004	Chicago	2002						x
Sailor & Lu 2004	San Francisco	2002						x
Sailor & Lu 2004	Philadelphia	2002						x
Sailor & Lu 2004	Atlanta	2002						x
Sailor & Lu 2004	Salt Lake City	2002						x
Masson et al., 2008	Toulouse	2004	x	x	x			
Pigeon 2007b	Toulouse	2004	x	x	x			x
Barzyk & Frederick 2008	Chicago	2005	x	x	x	x		
Lemonsu et al. 2008	Montréal	2005	x	x	x			
Christen et al. 2009	Vancouver	2008	x	x	x			
Moriwaki 2008	Tokyo	2008						x
Pearlmutter et al. 2009	Negev, Israel	2009	x	x	x	x		

1.6 Conclusions & Requirements

Theoretical knowledge on the processes forming the UEB and the resultant effects on the urban boundary layer is well developed. For typical urban areas, the daytime energy balance is characterized by a significant storage heat flux term, a strong sensible heat flux away from the surface, and weak evapotranspiration. As a consequence of strong nocturnal release of stored heat, both turbulent heat fluxes remain directed upward on average at night, a notable difference to the rural environment. This has consequences for the stability of the urban inertial sublayer and the roughness sublayer which are thermally unstable most of the time (Christen 2005). The crucial issue for researchers is the complex surface structure of urban areas which makes direct micrometeorological measurement and exact quantification of energy fluxes difficult. Modeling the UEB is not straightforward and even if several types of models are in use, and the computational power is enhanced, these include many simplifications and assumptions and it is challenging how they will be able to model the UEB in its whole complexity and for complete urban areas. Remote sensing techniques can cover large areas but are not able to account for all the individual terms of the UEB. Also, they usually have poor temporal resolution and other technical disadvantages. For each of the three fields of scientific research, measurement, modeling and remote sensing, more intensive investigations are required to increase the knowledge of the manifold urban influences on the energy balance.

One of the key requirements is attention to the closure of the urban energy balance. This implies an enhancement of understanding for each term of the UEB, especially:

- A better modeling of multiple radiation reflections in urban street canyons.
- Enhanced knowledge about the role of urban vegetation for evaporation, the influence of snow and the partitioning of the turbulent fluxes.
- Intensified investigations on the spatial and temporal pattern of the storage heat flux for different types of surfaces, its parameterization in models, and its determination with remote sensing techniques.
- Further integration of advection of heat and moisture into models and its consideration in UEB studies at sites where it might have a significant influence.
- A general increase in knowledge on the part of the anthropogenic heat flux in the UEB which implies the development of better methods to estimate it.
- Additional investigations on the importance of other sources and sinks of energy in the urban system.

Many of these requirements can be approached through:

- More field campaigns in varying cities with a focus on the UEB and on its single processes. For example, it has to be kept in mind that results from SEB investigations for non-European cities cannot straightforwardly be applied for European cities as land use, climate and urban metabolism probably will differ (Piringer 2002).
- A homogeneous framework for UEB studies with standardizations of methods and increased communication in the field of urban climate, for example as proposed by Oke (2005).
- Improved numerical models through better parameterizations and comparisons with measurements and other models in standardized frameworks, for example as conducted by Grimmond *et al*. (2007).
- Increased linkage of the different scales, especially in models. Connections established between mesoscale synoptic processes and those at the surface of a street canyon or rooftop and between. Increased resolution will improve the quality of the results.
- A better integration of the different investigation methods (measurement, modeling, remote sensing) into each other benefitting from the respective strengths and over coming weaknesses. This may be achieved by better communication between researchers originating from the fields of micrometeorology, numerical modeling and remote sensing.

Table 1.16: Four possible classifications for UEB models with sub-classes (after Grimmond *et al*. 2009a).

Classification Type	Sub-classes
Vegetation inclusion	a. Integrated: vegetation within tile that has the build facets so it can interact/respond to the exchanges associated with this layer. b. Separate Tile: vegetation and built parts of the surface are treated separately and do not interact until a layer above the surface scheme. The fluxes are a spatially weighted mean. c. None: assumed to be no vegetation present.
Urban land surface scheme layers resolved	a. Slab: represent the urban area in terms of a surface with appropriate thermal characteristics (e.g. Best 2005) b. Single layer: a layer of buildings with summed overall surface heat exchange (e.g. Masson 2000, Kusaka *et al*. 2001, Harman *et al*. 2004). c. Multi-layer: energy exchange at multiple levels within the canopy.
Facets and aspects resolved	a. Whole. Individual walls, roofs and roads are not resolved. b. Roof, Walls and roads are resolved but without orientation. No sunlit and shaded facets. c. Roof, Walls and roads are resolved with orientation. Sunlit and shaded facets exist.
Anthropogenic heat flux	a. Internal Temperature prescribed, used to calculate the other fluxes. b. Prescribed flux value c. Modelled all or components of the flux d. None: the flux is assumed to be 0 W m^{-2} or not to exist.

Table 1.17: Urban surface energy balance models. Acronyms, model names and publications with key details (from Grimmond *et al*. 2007)

CODE	Model Name	Reference with details of model
BEP02	Building Effect Parameterization	Martilli *et al*. (2002)
BEP05	Building Effect Parameterization	Hamdi (2005); Hamdi & Schayes (2005); Hamdi & Schayes (2007)
CAT	Canyon Air Temperature	Erell & Williamson (2006)
CLMU	Community Land Model -Urban	Oleson *et al*. (2008a,b)
ENVImet	Environmental Meteorology Model	Bruse & Fleer (1998)
GCTTC	Green Cluster Thermal Time Constant model	Shashua-Bar & Hoffman (2002; 2004)
HIM	Heat Island Model	Saitoh *et al*. (1996)
HIRLAM-U	Urbanised version of DMI-HIRLAM model	Baklanov *et al*. (2005, 2006); Mahura *et al*. (2006); Zilitinkevich *et al*. (2007)
LUMPS	Local-scale Urban Meteorological Parameterization Scheme	Grimmond & Oke (2002), Offerle *et al*. (2003)
MM5u	Penn State/NCAR Meso-scale Model model, where urban modifications have been incorporated	Dandou *et al*. (2005)
MOSES1T	Met. Office Surface Exchange Scheme 1 Tile	Best (2005); Essery *et al*. (2003)
MOSES2T	Met. Office Surface Exchange Scheme 2 Tile	Best *et al*. (2006); Essery *et al*. (2003)
MOUSES	Met Office Urban Surface Exchange Scheme	Harman *et al*. (2004a; 2004b)
MUCM	Multi-layer Urban Canopy Model	Kondo *et al*. (2005); Kondo & Liu (1998)
MUKLIMO	Microscale Urban Climate Model	Sievers (1995)
NSLUCM	Noah land surface model/Single-layer Urban Canopy Model	Kusaka *et al*. (2001); Chen *et al*. (2004)
PTEBU	Photovoltaic Town Energy Balance for an Urban Canopy	Tian *et al*. (2007)
R-AUSSSM	Revised Architecture-Urban-Soil-Simultaneous Simulation Model	Tanimoto *et al*. (2004)
RUM	Reading Urban Model	Harman & Belcher (2006)
SEBM	Surface Energy Balance Model	Tso *et al*. (1991)
SHIM	Surface Heat Island Model	Johnson *et al*. (1991)
SLUCM	Simple Single-layer Urban Canopy Model	Kusaka *et al*. (2001)
SM2U	Soil Model for Submesoscales, Urbanized Version	Dupont & Mestayer (2006); Dupont *et al*. (2006)
SUEB	Slab Urban Energy Balance Model	Fortuniak *et al*. (2004); Fortuniak *et al*. (2005)
SUMM	Simple Urban Energy Balance Model for Meso-Scale Simulation	Kanda *et al*. (2005a); Kanda *et al*. (2005b)
SUNBEEM	Simple Urban Neighbourhood Boundary Energy Exchange Model	Arnfield (2000)
TEB	Town Energy Balance	Masson (2000); Masson *et al*. (2002); Lemonsu *et al*. (2004)
TEB07	Town Energy Balance 07	Hamdi & Masson (2008)
TUF2D	Temperatures of Urban Facets in 2D	Krayenhoff & Voogt (2007)
TUF3D	Temperature of Urban Facets in 3D	Krayenhoff & Voogt (2007)
UCLM	Urban Canopy Layer Model	Mills (1997)
UCM	Urban Canyon Model	Sakakibara (1996)
UEB	Urban energy balance	Montávez *et al*. (2000)
UHSM	Urban Heat Storage Model	Bonacquisti *et al*. (2006)
VUCM	Vegetated Urban Canopy Model	Lee & Park (2007)

Part II: Water

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2.1 Introduction

In much of the world, urban growth puts pressure on already limited water resources (water extraction is greater than recharge in many regions of the world) and exerts a strain on ageing urban water infrastructure (Kjellen & McGranahan 1997, Lundin 1999). Compounding this are effects of human induced climate change (Maheepala & Perera 2003).

Many traditional urban water systems can be considered as non-sustainable due to the supply of high quality treated water for all processes (residential, industrial and irrigation), reliance on water for the transport of waste, and minimal re-use of waste water which typically is treated before being returned to natural water courses (Wilderer 2004). Therefore methods are required to improve levels of sustainability through improved water management both at and away from the point of demand. Such schemes could include the re-use/recycling of treated wastewater for non-drinking applications (e.g. irrigation), rainwater capture, toilet flushing and washing (Mitchell *et al*. 2001, Ragab *et al*. 2003b, Xiao *et al*. 2007). Additionally, the separation of solid waste using sceptic systems would reduce water use in the removal of sewage (Wilderer 2004). In order to determine the effectiveness of such methods an urban hydrological framework is required.

The urban environment is significantly different to natural hydrological watersheds[1] in terms of land use, water flows and surface cover leading to the modification of the hydrological cycle. In addition the transport and removal of water through the piped water system adds an anthropogenic component to the cycle. Artificial surfaces found in urban areas enhance the surface runoff (rate and volume) leading to an enhanced risk of flooding and the transport of pollutants (Burian *et al.* 2002), along with a reduction in infiltration leading to lower replenishment of groundwater (Stephenson 1994).

The complexity of the urban water system provides many challenges in the measurement and quantification of water usage. Therefore there is a need to define a suitable hydrological framework to meet these challenges. A framework first proposed by C. W. Thornthwaite in 1944 (Dunne & Leopold 1978) is the water balance. This was originally applied to rural drainage basins. Not until the 1970's did the water balance for urban catchments appear in the hydrological literature (Aston 1977, Lindh 1978).

The Urban Water Balance (UWB) applies the principle of mass conservation to the transfer of water through a specific domain or catchment[2] (Grimmond *et al*. 1986), allowing the study of both spatial and temporal patterns of water supply and usage (Mitchell *et al*. 2001). The UWB framework allows estimations of urban hydrologic processes which are intrinsically different to those observed in rural areas due to differences in land surface properties, the presence of artificial piped networks and human activities (Grimmond *et al*. 1986, Haase 2009).

Here, we present an overview of the UWB with particular focus on how it is measured and modelled, while identifying future research needs to improve the assessment of sustainable urban water practices. First, the urban water balance and the scales at which it is studied in a hydrological and atmospheric context are discussed. Then an overview of each component is considered. The methods used to measure and model each term are then discussed. The UWB studies that have been undertaken over the last thirty years are summarized and their results compared with an emphasis on scale. Whole UWB models are discussed before a review on future research required to fill gaps in scientific knowledge concerning the observation and modelling of the urban water balance.

[1] A natural watershed is the area drained by a river and its tributaries.
[2] An urban catchment may be defined by the natural drainage area or by the pipe networks present. Using the natural drainage may not result in the same area being defined by the sewer and stormflow pipe system (or combined sewers. Consideration also needs to be given to the location of pipes bringing water into the area.

2.2 The Hydrological view – The Urban Water Balance

The UWB is based on the conservation of mass (Grimmond & Oke 1991):

$$P+I+F = E+R+\Delta S+\Delta A \quad (2.1)$$

where P is precipitation, I is the urban piped water supply, F is water release due to human activity, E is evapotranspiration, R is runoff, ΔS is net change in water storage and ΔA is the net advection of water in and out of the catchment. Each of the terms is usually expressed as a depth of water (typically mm) per unit time (user determined but typically hour, day, month or year) or alternatively as a volume (m^3 per unit time) where the scale of the catchment or watershed is incorporated. It is also common to express individual terms as a percentage of the annual precipitation (often assumed to be the main input into the system) especially in the study of individual components such as runoff and evapotranspiration (e.g. Berthier *et al.* 2006, Xiao *et al.* 2007).

The UWB is linked directly to the surface energy balance as the mass of evapotranspiration, equivalent to the energy term for latent heat flux (Q_E) (Grimmond & Oke 1991):

$$Q_E = L_V E$$

where L_V is the latent heat of vaporisation. As noted in section 1, the surface energy balance is (Oke 1988):

$$Q^*+Q_F = Q_E+Q_H+\Delta Q_S \quad (\mathrm{W\ m^{-2}})$$

where Q^* is the net all-wave radiation, Q_F the anthropogenic heat flux, Q_H the turbulent sensible heat flux and ΔQ_S the net heat storage flux within the urban canopy.

2.3 Scales

The spatial and temporal scales used in the calculation, observation or modelling of each term of the UWB are important to consider. Timescales are particularly important due to the wide range of dynamical process scales associated with each term (Figure 2.1). Therefore a timescale needs to be utilised that in essence captures the majority of these scales for all processes, with the majority of studies using a daily time step (Van de Venn 1990).

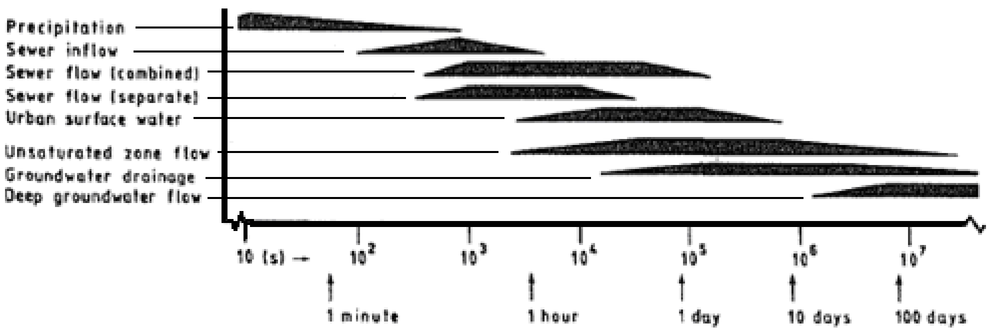

Figure 2.1: Timescale associated with terms n the urban water balance (Van de Venn 1990).

UWB studies can be organised into six typical spatial scales (Table 2.1). The smallest scale is an element which has its own individual properties. It is common for UWB studies to look at the relative land cover fraction of each element type before applying their individual properties to a larger scale (e.g. Pauliet & Duhme 2000, Rogriguez *et al*. 2008). While the largest scale tends to be regional in which studies focus on mixed land use types (e.g. Jia *et al*. 2001). Equally important are atmospheric scales as they impact spatial variation of precipitation, flow regimes and application of meteorological theory (e.g. Monin-Obukhov similarity theory). There are three main atmospheric scales that influence urban catchments; *viz*, micro, local and meso (Figure 1.1).

The UWB framework allows continuous observation of urban hydrologic processes instead of focusing on individual events (Cleugh *et al*. 2005) and can be used in modelling applications such as water use and management models (Neupane 2008). However, there is always a limitation to the number of measurements

available for the range of temporal and spatial scales and the complexity of the urban hydrological system. Therefore care is required when comparing monitoring and modelling studies to account for the scales and variables considered in the study as those chosen impact on the magnitude of the UWB.

Table 2.1: Typical spatial and meteorological scales used in urban water balance studies.

Name	Abbr.	Typical Area (km^2)	Meteorological Scale(s)	Description
Element	El	$< 1x10^{-5}$	Micro	Individual surface element with its own individual hydrological properties (e.g. trees, grass and pavement).
Property	Pr	10^{-5} to 10^{-1}	Micro	Individual property made up of one or more surface elements (e.g. residential house with garden).
Neighbourhood	Nh	10^{-1} to 10^{1}	Local	Group of properties and the infrastructure connecting them (e.g. roads and footpaths).
City	Ci	10^{1} to 10^{3}	Local	Whole city made up of numerous land use types, properties and surface elements.
Physical Catchment	Pc	10^{1} to $>10^{3}$	Local and/or Meso	Natural hydrological catchment in which urban elements are present.
Regional	Rg	$> 10^{3}$	Meso	Large area made up of multiple urban areas and physical catchments.

2.4 Precipitation (P)

Precipitation is a key input into the UWB as the amount and intensity directly impact the potential magnitude of evapotranspiration, runoff and infiltration, and the amount of recharge to surface and groundwater stores. The components of total precipitation (*P*) are:

$$P = P_r + P_h + P_s + P_m \quad \text{(mm per unit time)}$$

where P_r is rainfall, P_h is hail, P_s is snow and P_m is atmospheric moisture in which touch the surface in the form of fog, mist and/or dew. The form of precipitation dictates the timing of the availability of water for runoff, infiltration and evapotranspiration. Snow and hail, which fall in a solid/ semi-solid state, have to undergo a change of state to liquid or gaseous form and thus for a time period may be recorded as an increase in storage in the UWB. Snow storage, in particular, can last for many months in climates at higher latitudes adding complexity to the balance as the rate of melting (which in turn becomes runoff) needs to be determined and a change in the calculation of evapotranspiration rate is required (Semádeni-Davies & Bengtsson 1999, Lemonsu *et al*. 2008).

Dew (P_m) forms when the near surface temperature falls below the dewpoint temperature as a result of radiative cooling which in turn leads to condensation at the surface (AMS 2000). There have been very few studies into urban dew although enhanced moisture within the urban atmosphere as a result of anthropogenic activity coupled with strong radiative cooling certainly suggests potential for non-negligible accumulations. Work by Richards (2002) using scale models in Vancouver, Canada found that dew amounts were not insignificant and should be included in the urban water balance with values between 0.1 and 0.3 mm day^{-1} on nights with optimal conditions for dew formation.

Precipitation measurement within urban areas has traditionally used tipping bucket raingauges within meteorological enclosures (e.g. located at airports as part of the Automated Surface Observing System (ASOS), NOAA 1998) or other sites. It is widely accepted that point measurements do not afford sufficient spatial, and in certain circumstances, temporal resolution (Berne *et al*. 2004, Vaes *et al*. 2005). To address this problem a range of methods has been used (Table 2.2).

Despite their limitations raingauges are still important due to knowledge acquired over their long-term use for climate measurement and suitable temporal resolution. Studies have addressed this problem using a network of raingauges to account for spatial variability (e.g. Huff & Vogel 1978, Grimmond & Oke 1986) and for use in the correction of alternative rainfall detecting and modelling methods (Berne *et al*. 2004, Vieux & Bedient 2004, Segond *et al*. 2007).

Table 2.2: Examples of urban precipitation monitoring and modelling studies (by scale, see Table 1 for definitions).

Reference	Motivation	Location	Scale	Techniques & Major Findings
Huff & Vogel (1978)	Urban & topographic effects on precipitation.	METROMEX, St. Louis, U.S.A.	Rg	· Extensive gauge network (METROMEX). · Apparent enhancement downwind of urban areas.
Vaes *et al*. (2005)	Areal rainfall correction.	Flanders, Belgium	Rg	· Correction coeffi. dependent on intensity & duration of precipitation and catchment size. · Coeff. tested using a spatial rainfall generator.
Segond *et al*. (2007)	Spatio-temporal rainfall simulation.	Dalmuir, Glasgow, United Kingdom	Rg	· Model tested using gauge network. · Model data in agreement with observed patterns.
Bornstein & Lin (2000)	UHI and convective storms study	Atlanta, USA	Rg	· Used mesonet and NWS precip. measurements · Thunderstorm initiation due to convergence. · Bifurcation of storms approaching the city.
Berne *et al*. (2004)	Optimal spatio-temporal precip. resolution for urban hydrology.	HYDROMET, Marseille, France	Ci, Rg	· Used radar and gauge network. · Identified optimal spatio-temporal res. for a number of catchment sizes.
Szolgay *et al*. (2009)	Mapping approaches for max. daily rainfall	Hron River Basin, Slovakia	Pc	· Three interpolation methods compared
Vieux & Bedient (2004)	Storm reconstruction to test urban hydrologic prediction accuracy	Houston, Texas, U.S.A.	Pc	· Radar rainfall & hydrology Vflo™ model · Radar bias adjustment for accurate prediction. · Gauge correction reduces model & observed errors.
Zhang *et al*. (2009)	Urban expansion impacts on summer precipitation.	Beijing, China	Ci	· Climatology & mesoscale model study. · Precip. amount has reduced with urban expansion. · Model shows decreased λE & increased H.
Shepherd *et al*. (2002)	Satellite derived rainfall measurement & urban rainfall modification	Atlanta, USA (+ 5 other cities in southern USA).	Ci	· TRMM rainfall data utilised. · Incr. monthly rainfall downwind of & over cities. · Inferred increased rates as a result of UHI effects.
Tilford *et al*. (2002)	Radar adjusted rainfall and urban runoff modelling.	Bolton, Lancashire, England	Ci	· Radar derived rainfall correction: probabil. matching · Sewer flow simulation using Hydroworks. ·Radar rainfall for spatial variability in models
Ragab *et al*. (2003a)	Rooftop rainfall and interception measurement.	Oxfordshire, England	El, Pr, Nh	· Roof slope, orientation: flow effects interception. · Roof intercepts 62-93% of rainfall.
Richards (2002)	Quantification and modelling of urban dew and surface moisture.	Vancouver, Canada	El, Pr	· Scale model constructed. · Dew not negligible compared to other UWB terms. · Potential for dew collection.

Care needs to be taken when using a raingauge to consider measurement error due to its design (Nystuen *et al*. 1996), its location (Ragab *et al*. 2003a), the effects of wind and, turbulence (Nespor & Sevruk 1999) and performance during intense rainfall (Molini *et al*. 2005). In recent years, new sensors, for example a sensitive pan to measure precipitation (e.g. Vaisala WXT510; Vaisala Oyj 2006), have been deployed in urban areas as an alternative to traditional raingauges (Basara *et al*. 2009).

Radar can provide spatial information but cannot be used alone due to uncertainty in its accuracy (Berne *et al*. 2004, Vieux & Bedient 2004). These uncertainties stem from errors due to amongst others surface clutter, beam attenuation and measurement height relative to the surface. A complete overview of the limitations, merits and applications of radar in urban hydrology and water balance applications is given by Einfalt *et al*. (2004). Radar can be used for both rainfall and snow or other solid particles.

To generate relatively accurate spatial rainfall estimates from radar requires the application of correction factors to remove measurement biases. This is often achieved using measurements taken with raingauge networks (Tilford *et al*. 2002). Berne at al. (2004) utilised both radar and a raingauge network to determine optimal resolutions (both temporal and spatial) for rainfall data for use in urban hydrology studies. They concluded that for a study area of 1 km^2 data with temporal resolution of 3 minutes and spatial resolution of 2 km^2 is required, while for an area of 10 km^2 data with resolution of 5 minutes and 3 km^2 is needed. Radar has the potential to meet these resolution needs with spatial resolutions of 0.1 to 1 km (dependent on radar wavelength and

specifications) and a temporal resolution considerably shorter than that of raingauges. However, Einfalt *et al*. (2004) highlighted that care needs to be taken when averaging radar data due to it being an instantaneous volume measurement. A number of modelling studies (Table 2.2) have worked on correcting for spatial variability in raingauge network data through interpolation (Vaes *et al*. 2005, Segond *et al*. 2009) and determining radar derived rainfall errors and biases (Vieux & Bedient 2004).

Satellite measurement of rainfall using radar such as the Tropical Rainfall Measuring Mission (TRMM) is another method to observe rainfall over larger spatial scales of the order 250 km (Heymsfield *et al*. 2000). Shepherd *et al*. (2002) utilised TRMM during a study of rainfall modification over the Atlanta area. While the suggestion of rainfall modification continues to be controversial (Diem *et al*. 2004), the use of satellite data in the estimation of precipitation remains a potential tool.

2.5 Piped Water Supply (I)

The total piped water supply (I), an input into the UWB of water consists of:

$$I = I_U + I_R + I_G + I_S \quad \text{(mm)}$$

where I_U is the internal residential/industrial water use, I_R is water used for irrigation, I_M is grey or other reused water and I_S is the leakage to/from the piped network.

The magnitude of the water supplied is driven by a combination of demand from urban inhabitants and supply by the water utility companies or agencies, which is determined by availability of surface and groundwater supplies. When such supplies are low, water restrictions may be implemented which limit water use to essential needs. This in turn impacts other terms in the UWB (e.g. evapotranspiration due to reduced moisture availability). Measurement of the supplied water is often from water utility company water meters (e.g. Morris *et al*. 2007).

Often piped water supply in studies is considered in terms of internal and external components (Grimmond *et al.* 1986, Mitchell *et al*. 2001), with internal water being that used within buildings for human activities (e.g. drinking and sanitation), while external water is that used outdoors (e.g. irrigation).

Internal residential/industrial water usage (Table 2.3) is difficult to quantify accurately. Grimmond & Oke (1986) used a water meter mounted in the water supply pipe to a neighbourhood to determine an average daily water use of 827 L per household in Vancouver, Canada. Although most water utlilites have estimates of average consumption by appliances typically found in residences etc.

Irrigation is a major component of piped water use in urban areas where seasonal precipitation and weather patterns are particularly variable (Mitchell *et al.* 2001), with variability in irrigation related to specific weather events (Grimmond & Oke 1986). However, determining the actual amount of irrigation (as with other water usage) is a much more complex problem as it is related to the human perception and behaviour (Arnfield 2003). It has been shown that preventing over-irrigation or poor application can lead to a reduction in the amount of water required (Xiao *et al.* 2007). Typical irrigation values for private gardens were reported by Mitchell *et al*. (2001) to be of the order of 16 to 34% of total piped water. Measurement of irrigation can be inferred from the total water usage by assuming a constant base rate (Grimmond & Oke 1986) or through direct measurement by a water flow meter connected to the irrigation system (Xiao *et al*. 2007).

Through water management techniques, such as rain water collection and use of grey water, secondary water usage (e.g. irrigation and toilet flushing) from precipitation can provide the 18,250 L annually (approximately 30-40%) for toilet flushing of a typical household in the United Kingdom (Ragab *et al*. 2003a) and 9 % of the irrigation needs for a property in Los Angeles, California (Xiao *et al*. 2007).

Leakage to, and from, the piped supply network to the surrounding soil is common place where the ageing infrastructure was installed early in the 20th century (Tilford *et al*. 2002). The magnitude of this leakage impacts soil moisture. Estimates of leakage values vary from 3% of supplied water annually (Mitchell *et al*. 2003), to 10% (Morris *et al*. 2007), 28 % of the 8.66x10^{11} L consumed water for London (Best Foot Forward 2002).

An alternative to direct measurement is to construct a water use profile using categories (e.g. water used for washing, flushing the toilet and cooking) to determine water usage based on the number of inhabitants in each property (Mitchell *et al*. 2001). Gumbo (2000) used a combination of utility estimates and water use profiling to determine an average daily water use of 720 L in a suburb of Harare, Zimbabwe.

Table 2.3: Example of urban studies that report piped water supply (ordered by scale, largest to smallest).

Reference	Location	Area (km^2)	Scale	Methods	Annual Household Water Use (litres)	Irrigation (%)	I (mm)
Aston (1977)	Hong Kong	1046	Rg	Utility reported	2.83×10^5 (total)	-	1310
Semádeni-Davies & Bengtsson (1999)	Luleå, Sweden	29	Cw	Utility reported	1.09×10^5 (total)[1]	0	266
Mitchell *et al*. (2003)	Canberra, Australia	27	Pc	Water use profile & Aquacycle model (Mitchell *et al*. 2001)	1.22×10^5 (internal)[2] 8.49×10^4 (external)[2]	41	200
Gumbo (2000)	Harare, Zimbabwe	6.5	Pc	Water use survey and utility reported	2.10×10^5 (internal) 5.23×10^4 (external)	20	323
Grimmond & Oke (1986)	Vancouver, Canada	0.21	Nh	Single flow meter and resident survey.	3.02×10^5 (internal) 6.32×10^7 (external)	52	576
Neupane (2008)	Vancouver, Canada	0.21	Nh	Modelled	5.98×10^8 (total)[3]	-	614

[1] per capita based on a population of 71,000.
[2] based on population of 37,500 and 15,000 dwellings as reported in Mitchell *et al*. (2008).
[3] per household based on population of 420 and 191 dwellings.

2.6 Anthropogenic Water Release due to Combustion (F)

Anthropogenic water release due to combustion of fuels and from industry consists of:

$$F = F_M + F_I + F_C + F_W$$

where F_M is the release of moisture from air conditioning, heating and cooling applications, F_I is the moisture released from industry, F_C is the moisture released due to combustion of from vehicles and F_W is consumption of bottled water. This term has not been neither widely investigated nor often considered in UWB models (e.g. Grimmond *et al.* 1986, Mitchell *et al.* 2001).

Moriwaki *et al.* (2008), in one of the few studies, suggested that moisture release from air conditioning systems may explain the larger than expected latent heat flux measurements in Toyko. Their study developed a methodology to determine both anthropogenic water vapour and anthropogenic heating using energy consumption data and floor area for a number of building-use categories. During the study in Tokyo, anthropogenic water vapour fluxes showed seasonal variation with a peak during the summer in downtown areas with values exceeding 100 W m^{-2}. Suburban areas in comparison have modest fluxes of anthropogenic water (Moriwaki *et al*. 2008). The method did not consider emissions due to combustion from automobiles.

The input from F may be already included in the UWB if evapotranspiration is determined using the eddy covariance method (section 2.7). An additional anthropogenic source that needs to be considered is ‘bottled’ or packaged water, F_W. This water is imported from sources outside the urban area and used for drinking and cooking before eventually being released into the waste water system. The use of ‘bottled’ water is prevalent in a wide range of countries as it can be important where the piped water supply is not available to the individual property (e.g. Ouagadougou, Offerle et al. 2005) as well as being consumed as an alternative in those with well connected and well treated water. The estimated amount of ‘bottled’ water consumed annually in London alone was $94 \cdot 10^6$ L (Best Foot Forward 2002). Depending on the degree leakage of sewerage pipes this may or may play any role in the external water balance.

There is much further work required to determine the magnitude of anthropogenic water release and how both meteorological conditions and anthropogenic activity affect it. Both monitoring and modelling studies are required to determine the impact on other values in the UWB. Of particular interest is the modelling of evapotranspiration (section 2.7) where previous modelling studies have noted poor estimation of latent heat fluxes (e.g. Best et al. 2006) due to problems in representing available moisture in urban areas with little vegetation and the potential impact of climate change which with warming predicted within many cities around the world could lead to increased moisture release from air conditioning units (Day et al. 2009).

2.7 Evapotranspiration (E)

Evapotranspiration includes evaporation of surface water and transpiration through vegetation of water from the sub-surface vadose zone (Xiao *et al*. 2007). The term is used interchangeably with evaporation in many studies where it is impractical to separate the two components (Brutsaert 1982):

$$E = E_V + E_T$$

where E_V is evaporation and E_T is transpiration. Its energy equivalent, the latent heat flux, was discussed in section I.

It is often assumed that due to the high areal fraction of impervious surfaces within urban areas that evapotranspiration is not an important term in the UWB (especially in water engineering applications). However, it has been shown that evapotranspiration can be one of the most important terms in the UWB as a result of complex microclimates, surface storage and irrigation (Grimmond & Oke 1986, 1991, 1999, Mitchell *et al*. 2001, Berthier *et al*. 2006). For example, accounting for up to 38% of the annual water balance and 81% of the summer water balance observed in the temperate city of Vancouver, Canada (Grimmond & Oke 1986).

Urban parks are of particular interest due to the relatively high vegetation cover resulting in a distinct microclimate in comparison to surrounding more built up areas. The microclimate is akin to that of a desert oasis as an often limitless water supply (as a result of irrigation) coupled with higher temperatures (the urban heat island) result in elevated evapotranspiration rates. Spronken-Smith *et al*. (2000) in a park in Sacramento, USA observed that daily total evapotranspiration (measured using mini-lysimeters) in the park was greater than 300 % of the total from the surrounding irrigated suburban area.

At the micro-scale, two methods to determine evaporation are the evaporation pan (Aston 1977) and the lysimeter (Oke, 1979). The changes in the water level in evaporation pans provide an estimate of potential evaporation (energy limited rate). These simple sensors have the advantage of traditionally being available at a large number of standard climate stations, and the observations integrate a number factors that influence evaporation (wind speed, available energy, humidity and temperature) (Roderick *et al*. 2007). The lysimeter measures the water balance of an isolated volume of material, typically soil and vegetation located *in situ* in the ground (Grimmond et al. 1992). A time series of the weight of the mass with additional measurements of rainfall and runoff allow for evaporation to be determined. They have been used to study advection in lawns (Oke 1979) and urban parks (Sproken-Smith *et al*. (2000). Both evaporation pans and lysimeters have small spatial extent, which means a large number of samples need to be taken to be representative beyond the microscale.

At the neighbourhood, or local scale, micrometeorological techniques can be applied. Direct measurement of the latent heat flux, or evaporation, use eddy covariance methods through high frequency measurement of the vertical velocity (*w*) with a sonic anemometer and specific humidity (*q*) using a sensor (e.g. gas analyser such as a krypton hygrometer, Lyman-alpha, Infra-red gas analyser e.g. Licor 6262, 7000, 7500) (Grimmond 2006). Instruments positioned above the roughness sublayer are used to determine the covariance of the departures from the mean and therefore the flux:

$$E = \rho\, \overline{w'q'}$$

Care needs to be taken when using micrometeorological techniques to consider averaging time, the flux source area and sensor placement to ensure representativeness of the flux in an urban context (Schmid 1997, Grimmond 2006, Foken 2008). Eddy covariance measurements can be problematic during and immediately following precipitation. This is because of moisture influencing the transducers and gas analyser surface (Grimmond 2006).

A number of studies have been undertaken using micrometeorological measurements (Table 2.4). Based on field campaigns during the early 1990s in eight North American cities, E was observed using eddy covariance techniques. Grimmond and Oke (1999) found that areas with limited moisture (e.g. downtown and industrial areas) had limited water loss due to evaporation (< 1 mm per day), while those with irrigation (e.g. suburban areas) exhibited E losses up to 3 mm per day.

Over the last ten years or so there have been a number of field campaigns with observations of urban evapotranspiration (Tables 2.5, 2.6). In two European studies, Mestayer *et al*. (2005) during ESCOMPTE in Marseille, France, and Christen & Vogt (2008) during the BUBBLE (Rotach *et al*. 2005) campaign in Basel, Switzerland noted that observed latent heat fluxes of evaporation (Q_E) were generally low due to limited water availability during the period of observations (a period of limited precipitation and irrigation). Vegetation fraction within the urban area has an important impact on observed Q_E (Grimmond and Oke 1999, 2002, Christen & Vogt 2008). The magnitude of observed mean Q_E at three urban sites and one suburban site during BUBBLE ranged from 45 - 134 W m^{-2} during the day (compared with 251 – 282 W m^{-2} at surrounding rural sites) and 4 – 13 W m^{-2} during the night. Observations during ESCOMPTE were generally lower due to a much drier environment, with an average Q_E of 31 W m^{-2} during the day and 11W m^{-2} at night (Lemonsu *et al*. 2004). Observations remain, however, limited and more knowledge of the spatial variability of Q_E is needed.

In terms of temporal coverage, short term studies provide some valuable data but suffer because they are for a limited period. Long term urban measurements are required in a larger number of urban areas, with the results of such studies stored either in a dedicated database or communicated clearly in the literature. The latter would allow researchers to gain insight into the magnitude and climatology of urban evapotranspiration for use in future UWB monitoring and modelling studies. This is in part being addressed by the urban climatology community with the creation of an urban flux observation database - Urban Fluxnet which identifies study sites (http://www.kcl.ac.uk/projects/muhd/; www.urban-climate.org).

Deployment of micrometeorological instruments is not routine in the majority of urban areas and the eddy covariance technique has a number of limitations. Measurements tend to biased to dry conditions, and when the friction velocity is above a critical threshold. Unlike precipitation measurements they are not routinely made in many cities and there is the added complication of the need to locate the instruments on tall towers and undertake large amounts of data processing to determine the fluxes. Alternative methods of determination are to model the flux using one of a number of widely used equations (Table 2.7). These have typically been developed in non-urban areas so their appropriateness for use in urban areas needs to be assessed (Mitchell *et al.* 2001). The general approaches include:

a) The aerodynamic formulation for evaporation which requires knowledge of the humidity gradient between the surface and the atmosphere. There are a number of different ways to express the humidity gradient (e.g. vapour pressure (e), specific humidity (q), dew point temperature (T_d), absolute humidity or vapour density (ρ_V), mixing ratio (r)). Also needed is an expression of the ability of the atmosphere to transport the moisture away from the surface (e.g. eddy diffusivity for water vapour (K_v), surface resistance (r_s), surface conductance (g_s), surface resistance for heat). The aerodynamic method is used in a number of modelling applications (Table 2.5) (Noilhan & Planton 1989, Dupont *et al*. 2006):

$$E = \frac{\rho\delta(q_{sat}(T_s) - q_{ref})}{r_a}$$

Table 2.4: Examples of urban evaporation monitoring studies (by year of publication). [1] North American cities: part of the Multicity Urban Hydrometeorological Database (MUHD) ,include: Arcadia (Los Angeles), California; Chicago, Illinois; Mexico City, Mexico; Miami, Florida; Sacramento, California; San Gabriel (Los Angeles), California; Tucson, Arizona; and Vancouver, British Columbia, Canada.

Reference	Location	Scale	Description	Method	Results/Findings
Grimmond & Oke (1999)	Various North American cities[1]	Nh	Urban flux studies	EC	Urban E is an important flux. E> P in areas with high ext. water use
Spronken-Smith *et al*. (2000)	Sacramento, USA	El,Pr	Advection & Surface Energy Balance	Mini-Lysimeters	Daily total evaporation in irrigated park >300% than whole suburban area.
Grimmond *et al*. (2002)	Baltimore, USA	Nh	Suburban flux tower observations	EC	Latent heat flux reduced in autumn/winter due to leaf fall.
Ragab *et al*. (2003 a)	Oxfordshire, UK	El, Pr, Nh	Surface evaporation monitoring study.	Residual of water balance	Roof orientation, slope and prevailing wind effect evaporation. 14-38% of total precipitation evaporated.
Ragab *et al*. (2003 b)	Oxfordshire, UK	Nh	Surface evaporation monitoring study	Residual of water balance	24% of total precipitation was evaporated over the study area.
Mestayer *et al*. (2005)	Marseille, France	Ci	Urban surface energy balance study	EC	Spatial study of fluxes. Dry study area resulted in low observed E.
Christen & Vogt (2004)	BUBBLE, Basel, Switzerland	Rg, Nh	Multi-faceted urban boundary layer study	EC	Vegetation fraction impacts rate of evapotranspiration.
Hagishima *et al*. (2007)	Japan	El	Transpiration study of urban pot plants	Biomass weight	Isolated plants displayed a transpiration rate 2.7 x that of densely packed plants.
Lemonsu *et al*. (2008)	Montreal, Canada	Nh	Montreal Snow Experiment 2005	EC	Observed Q_E greater over snow than no snow (used latent heat of sublimation).
Vesala et al. (2008)	Helsinki, Finland	Nh	Urban flux studies	EC	Sector drastically different: vegetation & E

Table 2.5: Examples of urban eddy covariance studies and the processes observed (by year of publication) u – unpublished. LU land use. R: residential, I industrial, C Central Business District, M mixed (institutional, Park)

Reference	City	Code	Period	LU	Heat	Water	CO_2
Yap & Oke (1974)	Vancouver, Canada	V72	1972/186-244		Y	N	N
Cleugh & Oke (1986)	Vancouver, Canada	Vs83	1983/199-265	R	Y	N	N
Schmid et al. (1991)	Vancouver, Canada[7]	Vs86	1986/212-238	R	Y	N	N
Grimmond (1992)	Vancouver, Canada	Vs87	1987/021-179	R	Y	N	N
Oke *et al.* (1992)	Mexico City, Mexico	Me85	1985/034-090	R	Y	N	N
Grimmond et al. (1993)[1]	Sacramento, USA	S91	1991/231-241	R	Y	Y	N
Roth & Oke (1993)[1]	Vancouver, Canada	Vs89	1989/Jul	R	Y	Y	N
Grimmond et al (1994)[1]	Chicago, USA	C92	1992/Jul – 1993/Jun	R	Y	Y	N
Grimmond & Oke (1995)[1]	Tucson, USA	T90	1990/162-175	R	Y	Y	N
Grimmond & Oke (1995)[1]	Arcadia, USA	A93	1993/185-223	R	Y	Y	N
Grimmond et al. (1996)[1]	Arcadia, USA	A94	1994/187-206	R	Y	Y	N
Grimmond et al. (1996)[1]	San Gabriel, USA	Sg94	1994/188-206	R	Y	Y	N
Grimmond et al. (2002)[1]	Chicago, USA	C95	1995/165-222	R	Y	Y	Y[2]
Grimmond & Oke (1999c)[1]	Vancouver, Canada	Vl92	1992/223-238	I	Y	Y	N
Grimmond & Oke (1999c)[1]	Vancouver, Canada	Vs92	1992/206-261	R	Y	Y	N
Oke et al. (1999) [1]	Mexico City, Mexico	Me93	1993/334-341	C	Y	Y	N
Grimmond et al. (2002)	Baltimore, USA	Bm01	2001/Sep – ongoing	R	Y	Y	Y
Nemitz et al. (2002)	Edinburgh, Scotland	E99	2000/301-334,	C	Y	Y	Y
			1999/Oct-Nov		Y	Y	Y
			1999/May-Jun		Y	Y	Y
Spronken-Smith (2002)	Christchurch, New Zealand	Ch95	1995/223-236, 1996/29-49, 1997/197-221	R	Y	Y	N
Garcia-Cueto et al. (2003)[3]	Mexicali, Mexico	Mx01	2001/077-081		Y	Y	N
Roth & Satanarayana (2003[u])	Singapore	Si03	2003/077-092		Y	Y	Y
Soegaard & Moller-Jensen (2003)	Copenhagen, Denmark	Co01	2001-2002		Y	Y	Y
Christen & Vogt (2004)[4]	Basel, Switzerland[7]	Ba02	2002/161-191	R	Y	Y	Y[5]
Grimmond et al. (2004)	Oklahoma City, USA	Ok03	2003/178-213	R	Y	Y	N
Grimmond et al. (2004)	Marseille, France	Ma01	2001/155-197	C	Y	Y	Y
Miglietta et al. (2004)	Rome, Italy	Ro04	2004/Jan – ongoing	C	Y	Y	Y
Moriwaki & Kanda (2004)	Tokyo, Japan	Ty01	2001/May to 2002/Apr	R	Y	Y	Y
Walsh et al. (2004)	Vancouver, Canada	Vs01	2001/Aug to 2003/Jan	R	Y	Y	Y
Offerle et al. (2005)	Ouagadougou, Burkina Faso	Ou03	2003/039-051	R	Y	Y	N
Tejeda-Martinez & Jauregui (2005)	Mexico City, Mexico	Me98	1998/335-348		Y	Y	N
Velasco et al. (2005)	Mexico City, Mexico	Me03	2003/097-119		Y	Y	Y
Martensson et al. (2006)	Stockholm, Sweden	St02	2002/078-126		Y	N	N
Offerle et al. (2006)	Łódź, Poland[7]	Lo00	2000 – 2002	C	Y	Y	N
Coutts et al. (2007a,b)	Melbourne, Australia[7]	Mb04	2004/Feb -2005/Jun	R	Y	Y	Y
Newton et al. (2007)[1,6]	Miami, USA	Mi95	1995/132-173	R	Y	Y	N
Offerle et al. (2007)	Gothenburg, Sweden	Go03	2003/Jul to 2004/Aug	C	Y	N	N
Ramamurthy & Pardyjak (2007)	Salt Lake City, USA	Sl05	2005/Aug-Sep		Y	Y	Y
Lemonsu et al. (2008)	Montreal, Canada[7]	Mo05	2005/076-104	R	Y	Y	Y
Masson et al. (2008)	Toulouse, France[7]	To04	2004/Mar to 2005/Feb	C	Y	Y	N
Vesala et al. (2008)[8]	Helsinki, Finland	He05	2005/Dec – 2006/Aug	M	Y	Y	Y
Christen et al. (2009)	Vancouver, Canada[7]	Vs08	2008/190-241	R	Y	Y	Y
Sugawara & Narita (2009)	Tokyo, Japan	Tk98	1998/August		Y	N	N

[1] MUHD studies – information obtained from listed study reference and additional information from Grimmond & Oke (1999b; 1999c; 2002).
2. CO_2 results in Grimmond et al. (2002).
3. Reported in Roth (2007).
4. Study overview in Rotach et al. (2005)
5. CO_2 results in Vogt et al. (2006)
6. Additional information in Newton (1999).
7. Multiple sites in same urban area.
8. Further information in Jarvi et al. (2009)

Table 2.6: Moisture flux statistics from urban eddy covariance studies (by year of study). See Table 2.5 for site characteristics and references.

Code	Q_E Max (MJ d^{-1} m^{-2})	Q_E Mean (MJ d^{-1} m^{-2})	Q_E/Q^*	β	E Max (mm d^{-1})	E Mean (mm d^{-1})
Vs87		2.18-4.40	0.35-1.25	0.69-1.47		
T90u		4.90	0.39	1.58		1.99
S91u		4.38	0.45	1.14		1.77
C92			0.49	0.87		
Vl92		1.48	0.13	4.05		0.60
Vs92		2.68	0.30	2.72		0.60
A93		4.93	0.36	1.21		2.00
Me93	3.9	0.31	0.09	11.6		0.14
A94		4.70	0.30	1.60		1.94
Sg94		2.92	0.23	2.4		
C95		6.80	0.46	1.11		2.76
Ch95		3.32-5.40[1a]	0.25-0.41[1a]	1.99[1a]		
		0.41[1b]	0.10[1b]	-0.28[1b]		
		1.41-1.76[1c]	0.42-0.54[1c]	-1.12 – 0.39[1c]		
Mi95		4.58	0.33	1.47		1.87
Me98		1.8	0.13	3.8		
E99		2.0[2a]	-0.7[2a]	0.3[2a]		0.82[2a]
		1.6[2b]	-	1.1[2b]		-
		3.4[2c]	0.56[2c]	1.3[2c]		-
Lo00			0.38-1.34[3]	0.37- 1.45[4]		1.07
Ma01		3.3	0.23	4.02		
Mx01			0.07[5]	7.3[5]		
Ty01	15.8	2.58	0.26	2.39		
Ba02		8.6[6b]	0.21[6b]	2.28[6b]		
		7.6[6a]	0.18[6a]	2.62[6a]		
		3.9[6c]	0.14[6c]	4.27[6c]		
Ou03		1.8	0.23	2.5		
Mb04			0.19-0.30			
To04		1.2-2.1[7]	-28–0.49[7]	3.28-4.36[7]		
He05	13.0					
Mo05		1.8[8]	0.06[8]	6.30[8]		
Vs08		2.9[9a]	0.17[9a]	3.43[9a]		1.57[9b]
		3.8[9b]	0.25[9b]	1.86[9b]		

1 Range of daily average values noted due to differing flow conditions at two sites during different seasons (a) St Albans summer (1996/29-49), (b) St. Albans winter (1997/197-221), and (c) Beckenham winter (1995/223-236).
2 Values for three observation periods (a) 2000/301-334, (b) 1999/Oct-Nov, & (c)1999/May-June
3 Study calculated seasonal values so max (Oct to Apr) and min (Jun – Sep) are indicated.
4 Minimum value observed during Dec 2001 and maximum observed Jun-Sep 2002.
5 Clear sky daytime values from Roth (2007).
6 Observed values from daytime hours presented at three urban sites (a) Sperrstrasse, (b) Spalenring, & (c) Messe (observations at this site 2002/175-191).
7 Seasonal value range for the old city core of Toulouse presented (Masson et al. 2008).
8 Daytime average values.
9 Daytime values (10:00 to 16:00 LST) from two suburban sites (a) Sunset and (b) Oakridge.

where δ is the fraction of the surface saturated, q_{sat} the specific humidity at the surface, T_s the surface temperature and q_{ref} the specific humidity at a reference height. Or similarly (Nappo 1975; Carlson & Boland, 1978):

$$Q_E = -\frac{\rho L_v K_v M (q_a - q_0^*)}{\Delta z} \quad \text{where} \quad M \equiv \frac{E_0}{E_{op}} \equiv \frac{r_a}{r_a + r_s}$$

b) The energy balance formulation limits the evaporation rate based on the amount of energy that is available for the phase change of water.

c) Combination equations use the aerodynamic and energy balance formulations combined together. The most well known is the Penman-Monteith formulation. This forms the basis for the evaporation equations used by Grimmond et al. (1986), Grimmond and Oke (1991, 2002). One of the simplest methods is the Priestley and Taylor (1972) equation but it is for potential evaporation with the Priestley-Taylor α (typically 1.26):

$$E = \alpha \frac{s}{s+\gamma}(Q_* - \Delta Q_S)/L_v$$

where s is the slope of the saturation vapour pressure – temperature relation, γ is the psychrometric "constant". Only immediately following rainfall, or if the area is extensively wet (e.g. large lake), is this appropriate for use. This means that this simplification of the Penman-Monteith equation is not applicable most of the time and that actual evaporation needs to be calculated (Monteith 1981, Grimmond & Oke 1991):

$$E = \frac{s(Q_* + Q_F - \Delta Q_S) + {C_a V}/{r_a}}{L_v\left[s + \gamma(1 + {r_s}/{r_a})\right]}$$

where C_a the volumetric heat capacity of air, V the vapour pressure deficit of air, r_s the surface resistance and r_a the aerodynamic resistance.

d) The water balance formulations use some form of hydrologic balance (e.g. of the soil moisture content).
e) Empirical equations

Given that water is typically limited at the surface the actual evaporation rates are limited by surface controls and energy availability. When water is unlimited this is typically referred to as potential evaporation. This rate will typically be greater than the actual rate.

Within urban land surface schemes a number of approaches are used to determine urban evapotranspiration each with varying data requirements (see section I). Examples, include TEB (Masson 2000), TEB-ISBA (Lemonsu *et al*. 2007) and SM2-U (Dupont *et al*. 2006). Berthier *et al*. (2006), in their UHE parameterisation, calculates potential E with a modified Penman-Montieth method, and adjusts to actual evapotranspiration based on surface saturation.

Care needs to be taken when comparing modelling methods due to inherent differences in the methods utilised, forcings applied and the coefficients assigned. This was highlighted by Berthier *et al*. (2006) who compared two schemes (UHE and SM2-U) using the same data observed at a suburban field site at Rezé, Nantes, France. They determined that the annual evapotranspiration from UHE was 37% higher than that of SM2-U as a result of differences in commonly used variables and parameter uncertainty. A similar comparison was undertaken by Mitchell *et al*. (2008) who looked at the difference between the empirical linear evaporation scheme utilised in the UWB model Aquacycle (Mitchell *et al*. 2001) and the SUES model (Grimmond and Oke 1991). The results showed that despite two different approaches the model output was in reasonably good agreement (correlation coefficient (R^2) of 0.612 for data between 1989 and 1995). However, differences in mean monthly values as high as 23%. These differences highlight the need for further model comparison to understand these differences in addition to comparing performance with real urban observations.

An uncertainty in determining the magnitude of urban evapotranspiration is the magnitude of transpiration of urban vegetation (Mitchell *et al*. 2001). This was investigated in a study of urban potted plants by Hagishima *et al*. (2007) who found that isolated plants exhibited a greater transpiration rate than grouped plants suggesting that further modification to existing methods for determining urban evapotranspiration is required.

2.8 Runoff (R)

Runoff is the flow over the surface and through drainage pipes. It represents water that has not been captured by some intermediate store (e.g. tree canopy, roof or surface storage) or has not infiltrated into sub-surface stores within a particular time period. Urban areas typically have a greater fraction of impervious surfaces in comparison to rural areas (Shaw 1983); this leads to more rapid surface flows often enhanced by drainage networks (Semádeni-Davies & Bengtsson 1999). The increase in runoff can lead to higher probability of flooding and the transport of pollutants (Burian *et al*. 2002, Xiao *et al*. 2007).

The runoff consists of:

$$r = r_S + r_W + r_O + r_L + r_F$$

where r_S is storm water runoff (through storm drains), r_W is waste water flow (sewer system), r_O is runoff released by snow melt, r_L is surface ruoff (e.g. overland flow and roof runoff) and r_F is surface infiltration. The

rate and magnitude of runoff are regulated by the rate of precipitation, soil moisture content (influences infiltration), land surface properties (e.g. fraction of vegetation cover and permeability), local topography and the design of the drainage system infrastructure.

Table 2.7: Examples of urban evaporation/evapotranspiration models (by year of publication).

Reference	Location	Name	Description	E / Q_E Methods
Grimmond et al. 1986	Vancouver, Canada	UWB	Daily model based on Brustaert & Stricker	Modified combination model
Loudon Ross & Oke (1988)	Vancouver, Canada	Three urban energy balance models.	1-d surface energy balance models of varying complexity.	Bowen Ratio and specific humidity gradient method.
Grimmond & Oke (1991)	Vancouver, Canada	Single-source Urban Evapotranspiration Scheme (SUES)	Urban evapotranspiration – interception model.	Modified Penman-Monteith-Rutter-Shuttleworth scheme
Masson (2000)	-	Town Energy Budget (TEB) Scheme	Urban energy balance parameterization scheme.	Specific humidity gradient method
Mitchell *et al*. (2001)	Canberra, Australia	Aquacycle	Urban water balance model	Empirical linear relation f(potential E and surface moisture)
Grimmond & Oke (2002)	Numerous cities	Local-scale Urban Meteorological Parameterization Scheme (LUMPS)	Urban surface energy balance model	Partitioning approach using simplified Penman-Monteith and available energy
Lemonsu *et al*. (2004)	Marseille, France	TEB-ISBA (Town Energy Balance – Interactions between Soil, Biosphere & Atmosphere)	Combined urban and vegetation surface energy balance scheme.	Specific humidity gradient method
Berthier *et al*. (2006)	Rezé, Nantes, France	Urban Hydrological Element (UHE)	Mixed land use urban hydrological model	Penman-Monteith corrected with saturation extension indicator.
Claessens *et al*. (2006)	Ipswich R., MA, USA	-	Effect of land use change on E in a river basin.	Morton wet environment evapotranspiration method.
Dupont *et al*. (2006)	Rezé	Submesoscale Soil Model, urbanized version (SM2-U)	Urban mixed land-use parameterization scheme	Specific humidity gradient method.

There are a number of drainage system types in use in cities around the world. Dedicated sewer systems carry waste water from individual properties out of the catchment to treatment works. It is often assumed that the waste water (r_W) running through this system is equivalent to the household water usage from an individual property in what is described as the internal water system (Grimmond & Oke 1986, Mitchell *et al*. 2003). Most cities also have a dedicated storm water system to transport surface precipitation in an effort to reduce surface flooding. By keeping storm water separate from the sewer system minimises the risk of water contamination, however this does not account for pollutants washed from the surface. The storm water network and other processes that impact on the UWB and are not part of the wastewater or imported water systems are considered as the external water system.

In older cities there may be combined waste and storm water systems which use a single system for the transport of runoff from the study area. Tilford *et al*. (2002) noted that in a study in the northwest of England that combined systems have insufficient capacities to cope with heavy precipitation events leading to a risk of flooding and pollution transport. In addition to this, leakage from the system through deterioration of aging infrastructure impacts on the rate of runoff, contamination from pollutants and the magnitude of soil moisture surrounding the pipes. In Germany, Wolf *et al*. (2007) identified alone that approximately 20% of the nation's sewers suffer from leakage under certain flow conditions. Open channels such as streams and rivers which may have been altered (engineered) to deal with the additional runoff generated (Davenport et al., 2004).

It is often assumed that the majority, if not all, of the precipitation that falls on impervious surfaces results in runoff (Masson 2000, Rodriguez *et al.* 2000). This, however, is not the case as initially the surface is wetted (interception) which reduces amount of available water for runoff (Mansell & Rollet 2006). In addition to surface wetting the nature of the surface impacts the magnitude of runoff, for example infiltration through cracks (Lemonsu et al. 2007), and the slope of the surface (Mansell & Rollet 2006) impacts on infiltration.

Urban surfaces are not wholly uniform and have many cracks and joints in them through which water can infiltrate (Lemonsu *et al.* 2007). The onset of runoff is also delayed where imperfections in the surface are

wetted and is enhanced on sloped surfaces. Further work is required to determine a representative rate of urban infiltration (and for that matter runoff) as a whole and for individual surface types.

Urban runoff reduction through the use of water management techniques reduces the potential for urban flooding and the transport of pollutants. Methods utilised and tested include the use of water collection cisterns (Ragab *et al*., 2003a), water retention basins and gutters to slow the rate of runoff and promote infiltration (Xiao *et al.*, 2007), and green roofs which intercept rainfall before gradually releasing water to the drainage system (Bliss et al., 2009). A green roof study in Philadelphia, USA by Bliss *et al.* (2009) in which a flat roof was covered with small plants was shown to reduce runoff volume by up to 70% and to delay the peak flow rate in comparison with an adjacent control roof. Xiao et al. (2007) observed runoff reduction of 65% at a property scale through the use of a runoff collection drain and 12% by the creation of a runoff retention basin.

Direct measurement of runoff as a whole is particularly difficult due to its scale and the number of processes involved. Despite this a number of studies have measured components of the runoff equation (Table 2.8). Such measurements include the use of flow meters to determine discharge through the drainage network (suitable method for looking at loss of water through leakage by comparing readings) (Ragab *et al*. 2003b), soil moisture measurement (using moisture sensors and lysimeters) to determine soil water content (Xiao *et al*. 2007), runoff capture techniques for smaller study catchments (e.g. roof studies (Hollis & Ovenden 1988, Ragab *et al*. 2003a), property scale runoff (Xiao *et al*. 2007) and runoff flow measurement techniques on the boundaries of the study area (Stephenson 1994).

Hollis & Ovendon (1988) undertook an extensive stormwater runoff observation study for a number of urban surfaces. The techniques employed included the measurement of precipitation using rain gauges, soil moisture using tensiometers and flow measurement using potentiometric water level recorders located in runoff collection gullies and flumes. They determined runoff coefficients and depression storage for roofs, roads and car parks over the course of a year based on precipitatrion events. Results indicated that average percentage runoff for roads is 11.4% and 28.3% for rainfall less than 5 mm and greater than 5 mm respectively, while for roofs values of 56.9% and 90.4% were observed for each rainfall amount. Depression storage was determined using regression analysis of runoff with precipitation and through identification of rainfall events in which no runoff occurred. The event identification method produced higher values that the regression analysis for both roofs (0.52 mm and 0.42 mm) and roads (1.23 mm and 0.6 mm).

The majority studies determine runoff as a residual of the surface water balance approaches (e.g. Jia *et al*. 2001) or utilise infiltration/runoff coefficients (Hollis & Ovenden 1988). For example infiltration coefficients, were used by Pauliet & Duhme (2000) in Munich and Haase (2009) in Leipzig. Experiments for individual elements provide ratios that can be used. For example runoff-rainfall fractions observed by Mansell & Rollet (2006) for concrete and asphalt surfaces were in excess of 0.5; and for roofs (dependent on orientation relative to the prevailing wind and roof slope angle) average rates of 0.6-0.9 and 0.7 for road surfaces (Ragab *et al*. 2003 a,b).

Runoff is often modelled in UWB due to a lack of measured data and/or the size of the study catchment is large (Table 2.9). It has been modelled as a residual to the surface water balance (Rodriguez *et al*. 2000, Berthier *et al*. 2004):

$$r = P - E - r_F$$

where the modelling focus was on E and and infiltration rates. The infiltration rates may be modelled to land surface type specific coefficients (Zhang *et al*. 2009).

Using the runoff model L-THIA (Long Term Hydrologic Impact Assessment model) in Leeds (UK) Perry & Nawaz (2008) investigated impact of changes in fraction of impervious surfaces in gardens effect on runoff. They found the annual average runoff increased by 12% between 1971 and 2004 which has implications for potential flooding due to reduced infiltration.

In larger urban catchments two components models have been used: a land surface model to determine the amount of runoff and a drainage system flow to route this runoff through the study catchment (e.g. Rodriguez *et al*. 2000, 2008, Jia *et al.* 2000). Due to the wide variety of different surface types the land surface model may be applied to each to determine available water for runoff (Pauliet & Duhme 2000, Rodriguez *et al.* 2000).

2.9 Net Change in Storage (ΔS)

The net change in storage term refers to the change in water storage within the study catchment during the time interval of analysis. For long periods of time, such as a year, it is often assumed to be zero (i.e. no net-change). Its magnitude is determined by

$$\Delta S = \Delta S_g + \Delta S_m + \Delta S_W + \Delta S_A + \Delta S_n$$

where ΔS_g is the net change in ground water storage, ΔS_m is the net change in soil moisture storage, ΔS_W is surface water storage (e.g. ponds and lakes), ΔS_A is anthropogenic storage (e.g. storm water holding and water butts) and ΔS_n is the net change in snowpack storage.

Table 2.8: Examples of urban runoff studies (by year of publication).

Authors	Methods	Site	Scale	Key Findings
Hollis & Ovenden (1988)	Runoff capture	Hertfordshire, England	El	- Depression storage & surface moisture impacts on future runoff. - Rainfall amount & intensity have largest impact, E and seasonal variation impact limited.
Pauliet & Duhme (2000)	GIS, surface infiltration coefficients & rain gauge.	Munich, Germany	Cw	- Runoff impacted by infiltration rates. - Approx. 14% infiltration on built areas, 38% on natural surfaces & 23% of total precip is infiltrated.
Ragab *et al*. (2003 a)	Runoff capture	Oxfordshire, England	El, Pr, Nh	- Roof orientation & slope are main impacts on runoff.
Ragab *et al*. (2003 b)	Residual study. Soil moisture profiles, storm drain flow & rain gauge	Oxfordshire, England	El, Pr, Nh	- Residual method found runoff-rainfall ratio of 0.7. - Road surface infiltration of approx. 9%
Mansell & Rollet (2006)	Runoff capture and moisture profiling	Paisley, Scotland	El	- Majority of tested surfaces had runoff-rainfall rations > 0.5. - Infiltration limited on solid surfaces but significant through cracks (52% of precipitation). - Rougher surfaces had higher E rates.
Xiao *et al*. (2007)	Runoff capture	Los Angeles, U.S.A.	Pr	Water management methods tested reduced to street runoff significantly.
Bliss *et al*. (2009)	Runoff capture	Pittsburgh, USA	Pr	Green roof Runoff volume reduced by up to 70% in comparison with control.

Table 2.9: Examples of urban runoff models (by year of publication).

Reference	Name	Location	Process	Methods
Rodriguez *et al*. (2000)	Semi-Urbanized Runoff Flows (SURF)	Rezé, Nantes, Neighbourhood	Urban runoff model	GIS/DEM land use model. Simplified water budget (r=P-E-Infiltration)
Burian *et al*. (2002)	Storm Water Management Model (SWMM)	Santa Monica Bay and Los Angeles, California	Storm water runoff model.	Deterministic model utilising precip, catchment characteristics and storm drain infrastructure.
Tilford *et al*. (2002)	Hydroworks	Bolton, England	Combined sewer-storm water flow	Hydrol. model utilises sewer system information, precip from radar.
Berthier *et al*. (2004)	Urban Hydrological Element (UHE)	Rezé, Nantes, Neighbourhood	Urban runoff	2D numerical runoff model with a fine resolution soil grid.
Gash *et al*. (2008)	None given	Oxfordshire, England	Urban roof runoff (based on a forest evaporation model).	Empirical model based on an analogy between roof and tree canopy interception.
Mohrlok *et al*. (2008)	UL_FLOW	Rastatt, Germany	Analytical infiltration model	Timestep based steady-state effective infiltration model
Perry & Nawaz (2008)	L-THIA	Leeds, England	Hydrological impact assessment	Model analyses effect of land cover change on runoff.
Wang *et al*. (2008)	Urban Forest Effects-Hydrology model (UFORE-Hydro)	Baltimore, Maryland, USA. Catchment	Tree effects on urban hydrology	Canopy and interception model. E based on Penman-Monteith. Infiltration & runoff uses TOPMODEL.

One of the main stores of water within an area may be the groundwater within the soil and deeper aquifer. The storage capacity is determined by the underlying geology and the type and structure of the soil which both impact the movement of ground water. Techniques to measure soil moisture include tensionmeters (Berthier *et al*. 2004), gravimetric sampling (Grimmond & Oke 1986), time domain reflectometry; and for groundwater levels can be observed through borehole (Stephenson 1994).

The recharging of groundwater is altered by reduced infiltration, leakage from the water supply and storm water pipe networks (Lerner 1990), and the altered evaporation rates. The infiltration rates are controlled by precipitation rate, surface storage, soil structure and moisture content in the vadose and groundwater level (Xiao *et al.* 2007, Mohrlok *et al.* 2008). Quantification of infiltration within soils is difficult due to non-linearity of movement with depth but estimates are often made using soil moisture profiles (Xiao *et al.* 2007), tracer studies (Barrett *et al*. 1999), infiltrometers (Lemonsu *et al*. 2007) and infiltration modelling (Mohrlok *et al*. 2008).

Road surface infiltration may also play a role either directly through the surface or via cracks and joints (Lemonsu *et al*. 2007). Traditionally infiltration through 'impervious' surfaces is not considered; however, a number of studies have shown that infiltration rates are significant and dependent on surface type and structure with rates ranging between from 6 to 30% of total precipitation (Ragab *et al*. 2003b, Berthier *et al*. 2004).

Many use a modified version of Green and Ampt's (1911) equation with a multilayered profile and assuming infiltration rate is at least equal to the precipitation rate (i.e. surface ponding occurs) (e.g. Jia *et al*. 2001, Xiao *et al*. 2007). It is limited by being applicable to natural surfaces. Mohrlok *et al.* (2008) created a 1-d model, UL_FLOW, based on a constant infiltration rate determined by the Urban Volume and Quality Model (Mitchell & Diaper 2005).

2.10 Net Moisture Advection (ΔA)

The net moisture advection is the horizontal transport of moisture by atmospheric flow. Scale or study domain definition (e.g. control volume or surface) is a fundamental consideration to assess if this term needs to be formally assessed. It is driven by flows at a number of atmospheric scales ranging from micro and local scale turbulence to mesoscale circulations (e.g. sea breezes and valley flow). In many UWB studies the net moisture advection is not considered (e.g. Grimmond *et al*. 1986, Lemonsu *et al*. 2007).

Pigeon *et al*. (2007) found during ESCOMPTE (an intensive urban measurement campaign) in Marseille, France (Mestayer *et al*. 2005) the sea breeze circulation reduced measured moisture at the study site due to horizontal advection. At the microscale, advection influences the amount of evapotranspiration by increasing its rate, this was observed in studies over irrigated lawns and parkland where edge and oasis effects are evident (Oke 1979, Spronken-Smith *et al*. 2000).

Measurements of advection are typically only undertaken during intensive field campaigns when a wide range of micrometeorology equipment may be available (Spronken-Smith *et al*. 2000, Pigeon *et al*. 2007). Evaluation of moisture advection can be undertaken using models; for example, using a mesoscale atmospheric model (Meso-NH) coupled with TEB-ISBA (Pigeon *et al*. 2007).

2.11 Past Urban Water Balance Studies

Over the last 40 years or so there have been a limited number of UWB studies (Table 2.10) with the earliest work in the 1960's (Bell 1972). Work continued throughout the 1970's (e.g. L'vovich & Chernogayeva 1977; Lindh 1978) and 1980's (e.g. Grimmond & Oke 1986; Stephenson 1994) with research focused on urban processes and the application of the water balance to urban areas. In the 1990's there were limited studies reported (Semádeni-Davies & Bengtsson 1999; Gumbo 2000). The 21st century with its apparent focus on urban sustainability and the potential effects of climate change has lead to a large number of UWB studies which focus on water sustainability and management techniques (e.g. Mitchell *et al.* 2001; Wolf *et al.* 2007).

There have been a number of changes in the techniques utilised in UWB studies to measure and model variables over time (Table 2.11). The early UWB studies (e.g. L'vovich & Chernogayeva 1977, Lindh 1978, Van de Ven 1990) had limited observations (typically precipitation, piped water supply, evapotranspiration and runoff with groundwater occasionally included) made with traditional techniques. The little modelling was based on

empirical relations and the calculation of residual values. This was in part due to the level of scientific understanding and technology available at that time, along with focused research aims.

With time, the range of measurements and methods utilised has changed with improved scientific understanding of processes, the advent of newer technology (e.g. the move from profile to eddy covariance techniques in micrometeorology), wider reaching research projects leading to a greater number of variables measured (e.g. the division of runoff into components) and the development of models for simulating processes from basic field measurements (e.g. modelling of evapotranspiration and soil water infiltration).

A summary of observed values (Table 2.12) and annual results (Table 2.13) show distinct differences between studies. This is a result of each UWB study having a unique spatial scale, geographical location, population dynamics, urban fabric, local climate and combination of modelling and monitoring techniques. This makes direct comparison of individual studies with others of limited use without a prescribed framework that takes into account these differences. The traditional method utilised to make comparisons while taking into account differences in study area is to determine the relative percentage of each term in the UWB equation (Table 2.13). While this has its merits, in that one can estimate the effects of changes in particular variables (e.g. precipitation) on the UWB for a particular location, it is limited (in the absence of a common set of fluxes) by differences in the number and type of measurements made in each study.

At the micro scale there have been property (parcel, lot) or smaller areas investigated; for example, car park and a small housing area (Van de Ven, 1990, Xiao *et al.*, 2007). At this scale spatial variability may be a less significant but measurement errors can still be significant (in excess of 20%, e.g. Van den Ven 1990). Clear definition of study area boundaries should help to reduce errors.

The impact of storm water runoff management techniques have been tested using two similar suburban properties (a study and control) (Xiao *et al.*, 2007). Techniques applied included rain water collection from roof guttering, construction of a water retention basin on the lawn and a driveway drainage channel placed across the driveway, with the latter two designed to promote gradual infiltration. Together these contributed to a near total reduction of runoff to the street from the property during a number of storm events.

Local scale studies, of neighbourhoods or small suburban areas, have focused on measurement of the UWB (Louden and Oke, 1986); impact of irrigation on evaporation (Grimmond *et al*., 1986; Grimmond and Oke, 1986); evaluation of water management techniques such as the reduction of storm water runoff, the collection and use of rainwater, recycling of grey water for low grade water use (e.g. sub-surface irrigation), the treatment of waste water on site (e.g. sceptic tanks) and controlled irrigation systems (Mitchell *et al.* , 2001, 2003, 2008); impact of urbanisation on ground water using data collected in parkland in Perth, Australia (McFarlane 1985); and suburban – natural grassland comparisons in South Africa (Stephenson, 1994).

At the city scale studies have been conducted to assess a variety of issues: future requirements for the removal of sewage in Sydney (Bell 1972); predictions for future water usage in Hong Kong (Aston, 1977); urbanization effects the water balance for Moscow (L'vovich and Chernogayeva, 1977); effects of snow cover and storage (Semádeni-Davies and Bengtsson; 1999, Luleå, Sweden); and performance and effectiveness of the water and sewerage system (Gumbo 2000; Harare, Zimbabwe).

At the meso-scale, the city has been studied in relation to its larger regional contect: to look at the role of water as part of an ecosystem study in Mexico City (Campbell 1982), cities in relation to their national context (Lindh, 1978, e.g. Sweden).

In 1990, Van de Ven presented results from an extensive year long conducted in the late 1970's of Lund, Sweden. He argued that measurement should be the focus of studies as model parameters and variables suffer from representativeness and transferability issues due to differences in scale. His caveat is valid to a point; however, it is not conclusive as both measurements and models suffer from these issues, therefore a combination of modelling and monitoring is required over a range of sites to address the above issues.

Projects since 2000 have focused on the effects of urbanisation and sustainable water management practices. Haase (2009) utilised the UWB method to investigate the long term effects of urbanisation (in particular changes in impervious land cover) on evapotranspiration, runoff and groundwater recharge for Leipzig, Germany. Lekkas *et al.* (2008) applied the Aquacycle model (Mitchell *et al.* 2001) to investigate water usage

and the performance of the drainage system (both storm water and wastewater) using measurements for Athens, Greece.

The European Union funded study, Assessing and Improving Sustainability of Urban Water Resources and Systems (AISUWRS, 2002-2005), focused on methods to improve sustainability of urban water and reduction of groundwater contamination (Wolf *et al*. 2007). Case studies to test the models and methods proposed including UWB studies in (Wolf *et al*. 2007): Doncaster, England (Morris *et al.* 2007); Rastatt, Germany (Klinger *et al.* 2007); Ljubljana, Slovenia (Souvent *et al.* 2007); and Mount Gambier, Australia (Cook *et al.* 2007).

Table 2.10: Urban Water Balance studies by study area size (some cited by Grimmond & Oke 1986, Stephenson 1994, Mitchell et al. 2003).u- unpublished

Reference	Location	Area (km^2)	Scale	Observation Period	Purpose
Lindh (1978)[1]	Sweden	4024	Rg	Annual, circa 1970	Research program
Aston (1977)	Hong Kong	1046	Rg	Annual 1971	Future water requirements
Bell (1972)	Sydney, Australia	1035	Ci	Annual 1962-1971	Future sewerage disposal requirements
L'vovich & Chernogayeva (1977)	Moscow, Russia	879	Ci	Annual	Determine influence of urbanization
Campbell (1982)	Mexico City,	?	Ci	Annual 1980	Ecosystem study
Lekkas *et al*. (2008)	Athens, Greece	357	Ci	Annual 2001	Study impacts of water management techniques
Haase (2009)	Leipzig, Germany	?	Ci	Annual 2007	Study into the effects of urbanisation on the water balance.
Semádeni-Davies & Bengtsson (1999)	Luleå, Sweden	29	Ci	Monthly, annual. 1992 – 1996	Investigating the effects of snow cover on the UWB
Wolf *et al*. (2007)	Mt Gambier, Australia	27	Pc	Annual	Water balance study investigating
Mitchell *et al*., (2003)	Canberra, Australia	27	Pc	Seasonal, annual. 1978-96	Quantification of urban water a balance continuous daily model (Aquacycle)
Van de Ven (1990)	Lund, Sweden	19.4	Ci	Annual 1978-1979	Water balance studies
McFarlane (1985)	Perth, Australia	?	Nh	Annual	Effect of urbanisation on ground water
Gumbo (2000)	Harare, Zimbabwe	6.5	Nh	Annual, monthly	Assessing urban water management using the urban water balance
Morris *et al*. (2007)	Doncaster, England	6.3	Ci	Annual 1997 (reference year)	Water balance study: management techniques on groundwater recharge below urban areas.
Wolf *et al*. (2007)	Ljubljana, Slovenia	0.76	Nh	Annual 2004	Study of a layered aquifer system and the effects of drainage infrastructure.
Stephenson (1994)	Sunninghill, South Africa	0.76	Nh	Annual 1987- 91	Water balance comparison (suburban)
Louden & Oke (1986[u])	Vancouver, Canada	0.21	Nh	Daily 2 summer months. 1980	Summer suburban water balance
Grimmond & Oke (1986)	Vancouver	0.21	Nh	Daily, monthly, seasonal, annual. 1982	Components in suburban water balance.
Van de Ven (1990)	Lelystad, Netherlands	0.2	Pr	Annual 1970-1984	Water balance study
Van de Ven (1990)	Lelystad	0.0076	El/Pr	Annual 1970 - 1984	Water balance study
Xiao *et al*. (2007)	Los Angeles	0.0007	Pr	Monthly, Annual. 2001-02	Water management practices to reduce urban storm runoff.

Table 2.11: Methods used for determining terms in UWB studies. Each method is indicated by the following: - = not measured in the study, X = unknown/not-defined method, Est = estimated (method unknown), Obs = observed using a gauge (precipitation & runoff), Uti = utility company reports/measurements, Res = residual (using direct calculation or simple relation), Mea = measured (method unknown), MoP = measured evapotranspiration using Penman-Monteith/Combination method, MoE = Modelled using an empirical technique (different types), MoB = modelled evapotranspiration using Bowen Ration method, Mod = modelled by unknown/un-defined method, MoF = modelled/simulated runoff/flow model, Cal = calculated using a unknown/un-defined method.

Reference	Area (km^2)	Measurement Techniques by Variable												
		P	***I***	***E***	***r***	ΔS	r_s	r_w	r_L	***W***	ΔW	ΔS_n	W_S	ΔS_m
Lindh (1978)	4024	Est	Est	Est	Est	-	-	-	-	-	Est	-	-	-
Aston (1977)	1046	Obs	Uti	Est	Est	-	-	-	-	Res	-	-	-	-
Bell (1972)	1035	Obs	Est	MoP	MoE	-	-	-	-	Est	-	-	-	-
L'vovich & Chernogayeva (1977)	879	Obs	-	Res	Mod	-	-	-	-	-	-	-	-	-
Campbell (1982)	?	Est	Est	Est	Est	-	-	-	-	-	-	-	-	-
Lekkas *et al*. (2008)	357	-	Uti	-	-	-	Est	Est	-	-	-	-	-	-
Semádeni-Davies & Bengtsson (1999)	29	Obs	Uti	MoE	-	-	Est	Mod	Est	Res	-	Est	Res	Res
Wolf *et al*. (2007)[1]	27	X	X	X	-	-	X	X	-	-	X	-	X	X
Mitchell *et al*., (2003)	27	Obs	Uti	MoE	-	Res	Obs	Obs	-	-	-	-	-	-
Van de Ven (1990)	19.4	Mea	Mea	Cal	-	-	Mea	Mea	-	Cal	-	Mea	Cal	Mea
McFarlane (1985)	?	X	X	X	-	-	X	X	-	X	X	-	-	-
Gumbo (2000)	6.5	Obs	Uti	-	-	-	MoF	Est	-	-	-	-	-	-
Morris *et al*. (2007)	6.3	Mea	Uti	MoE	-	-	Obs	Uti	-	-	Mod	-	-	Mod
Wolf *et al*. (2007)[2]	0.76	X	X	X	-	-	X	X	-	-	X	-	X	X
Stephenson (1994)	0.76	Obs	Uti	Cal	-	-	Obs	Obs	-	-	Obs	-	-	-
Louden & Oke (1986, unpub)	0.21	Obs	Obs	MoB/ MoE	Est	Est	-	-	-	-	-	-	-	-
Grimmond & Oke (1986)	0.21	Obs	Obs	MoP	Mod	Mod	-	-	-	-	-	-	-	-
Van de Ven (1990)	0.2	Mea	Mea	Cal	-	-	Mea	Mea	Mea	Mea	-	-	Cal	Mea
Van de Ven (1990)	0.0076	Mea	-	Cal	-	-	Mea	-	Mea	Mea	-	-	-	Mea
Xiao *et al.* (2007) Treatment site	0.0007	Obs	Obs	-	Obs	-	-	-	-	-	-	-	-	-
Xiao *et al.* (2007) Control site	0.0007	Obs	Obs	-	Obs	-	-	-	-	-	-	-	-	-

[1] Mount Gambier, Australia; [2] Ljubljana, Slovenia.

Table 2.12: Urban water balance study results (from largest to smallest study area). Missing values in the table represent variables that were not studied in each study.

			Measured / Modelled Variables (mm)												
Reference	**Area (km^2)**	**Period**	P	I	E	r	ΔS	r_s	r_w	r_L	W	ΔW	ΔS_n	W_S	ΔS_m
Lindh (1978)	4024	annual	701	235	360	576	-	-	-	-	-	0	-	-	-
Aston (1977)	1046	annual	1912	1310	1128	2158	-	-	-	-	64	-	-	-	-
Bell (1972)	1035	annual	1150	333	736	763	-	-	-	-	16	-	-	-	-
L'vovich & Chernogayeva (1977)	879	annual	700	-	400	300	-	-	-	-	-	-	-	-	-
Lekkas *et al*. (2008)	357	annual	-	1106	-	-	-	314	669	-	-	-	-	-	-
Semádeni-Davies & Bengtsson (1999)	29	annual	524	266	136	-	-	200	337	158	18	-	3	62	0
Wolf *et al*. (2007)	27	annual	703	101	327	-	-	225	90	-	-	364	-	10	18
Mitchell *et al*., (2003)	27	annual	630	200	508	-	1	203	118	-	-	-	-	-	-
Van de Ven (1990)	19.4	annual	662	427	298	-	-	230	893	-	72	-	7	421	10
McFarlane (1985)	?	annual	788	285	766	-	-	104	454	-	96	117	-	-	-
Gumbo (2000)	6.5	annual	820	323	-	-	-	166	343	-	-	-	-	-	-
Morris *et al*. (2007)	6.3	annual	578	220	314	-	-	101	169	-	-	212	-	-	2
Wolf *et al*. (2007)	0.76	annual	1091	1347	240	-	-	660	1084	-	-	497	-	255	16
Stephenson (1994)	0.76	annual	724	114	457	-	-	107	95	-	-	180	-	-	-
Louden & Oke (1986u)	0.21	summer	90	172	190	97	-25	-	-	-	-	-	-	-	-
Grimmond & Oke (1986)	0.21	annual	1215	576	578	1210	3	-	-	-	-	-	-	-	-
Van de Ven (1990)	0.2	annual	743	529	312	-	-	367	456	168	108	-	-	73	3
Van de Ven (1990)	0.0076	annual	777	-	138	-	-	396	-	340	106	-	-	-	9
Xiao *et al.* (2007) Treatment site	0.0007	annual	427	589	-	5	-	-	-	-	-	-	-	-	-
Xiao *et al.* (2007) Control site	0.0007	annual	427	547	-	227	-	-	-	-	-	-	-	-	-

Table 2.13: Summary of selected urban water balance equations used and the results from each study (from largest to smallest study area). Rounding errors maybe present in the calculations. *This study did not balance and there was insufficient information to determine why.

			Measured / Modelled Variables (%)												
Reference	**Area (km^2)**	**Period**	P	I	E	r	ΔS	r_s	r_w	r_L	W	ΔW	ΔS_n	W_S	ΔS_m
Lindh (1978)	4024	annual	75	25	38	62	-	-	-	-	-	0	-	-	-
Aston (1977)	1046	annual	58	40	34	66	-	-	-	-	2	-	-	-	-
Bell (1972)	1035	annual	77	22	49	51	-	-	-	-	1	-	-	-	-
L'vovich & Chernogayeva (1977)	879	annual	100	-	57	43	-	-	-	-	-	-	-	-	-
Semádeni-Davies & Bengtsson (1999)	29	annual	66	34	17	-	-	25	34	20	2	-	1	8	-
Mitchell *et al*., (2003)	27	annual	76	24	61	-	1	24	14	-	-	-	-	-	-
Van de Ven (1990)	19.4	annual	61	39	27	-	-	21	82	-	7	-	1	39	1
Morris *et al*. (2007)	6.3	annual	72	28	39	-	-	13	21	-	-	27	-	-	0
Stephenson (1994)	0.76	annual	86	14	55	-	-	13	11	-	-	21	-	-	-
Grimmond & Oke (1986)	0.21	annual	68	32	32	68	0	-	-	-	-	-	-	-	-
Van de Ven (1990)	0.2	annual	58	42	24	-	-	29	36	13	8	-	-	6	0
Van de Ven (1990)	0.0076	annual	100	-	18	-	-	51	-	44	14	-	-	-	1

2.12 Future Urban Water Balance studies: Sustainability of urban water resources

Comparisons require methods utilised and variables observed to be reported. Similarly, the representativeness of observations relative to the size of the study area needs to be assessed; in particular, a single point measurement of precipitation to represent a city, it likely not appropriate. Future studies, in the planning stages need to consider what measurements are required to ensure that meteorological and hydrological variables are representative of a particular study area.

In addition, a framework is needed to enable comparisons to be drawn between different case studies (both in terms of techniques used and magnitude of variables observed) while allowing improvements in the analysis of sustainable water practices in a range of cities across the world using reported results. These disparities are highlighted in Tables 2.11 and 2.13 which provide a summary of the measurement and modelling techniques utilised by each study and annual values observed, respectively.

For a number of studies the values presented are based on a single year's observation (e.g. Aston 1971, Gumbo 2000, Morris *et al*. 2007). This does not enable the determination as to whether the results are 'typical' in the context of the local climate, therefore there is the need for multi-year studies (e.g. Van de Ven 1990, Semádeni-Davies and Bengtsson 1999) to construct UWB climatologies. This needs to include the seasonal characteristics of the UWB due to distinct variations in precipitation and piped water use patterns apparent in studies that report them (Grimmond and Oke 1986, Mitchell *et al.* 2001). It is therefore imperative that annual and seasonal climate is considered when assessing potential sustainable water management practices.

2.13 Modelling the Urban Water Balance

The accurate representation of the UWB through modelling is imperative for the assessment of future sustainable urban water management practices, realistic simulation of urban surface processes and for predicting the effects of climate change. There are three general types of UWB models each with varying degrees of complexity and spatial extent (Table 2.14).

2.13.1 Mass balance

The first type of model is dedicated to the determination of the UWB for use in urban hydrology and water management technique assessment applications. These models utilise the mass balance based approach used by Grimmond *et al*. (1986) in their Urban Water Balance Model and combine both natural and anthropogenic hydrological systems.

The Urban Water Balance model (Grimmond *et al.* 1986) utilises daily meteorological data (e.g. net all wave radiation and total precipitation) and water use. Parameters required include physical land cover properties (e.g. fraction of impervious land cover and aerodynamic roughness length), surface and subsurface hydrologic properties (e.g. soil storage capacity), and initial conditions in terms of storage in the study area (e.g. soil moisture state). The model was demonstrated and evaluated using observations from Vancouver, Canada (Grimmond and Oke 1986) and suggested uses include investigation into urban irrigation and the urban energy budget (through the link with evapotranspiration).

Two urban water balance models developed in Australia based on the assessment of water management techniques are Aquacycle (Mitchell *et al*. 2001) and the Urban Volume and Quality model, UVQ (Mitchell and Diaper 2005). UVQ is essentially an expanded version of Aquacycle with the added ability to model contaminant fluxes and was developed for an urban water resource modelling toolbox during AISUWRS (Diaper and Mitchell 2007). Both models are based on the mass balance principle and are formed of inputs, outputs, flows and stores. Site specific input values are required to calibrate and run the models with three nested spatial scales in each (unit block (property), cluster (neighbourhood) and the study area as a whole (Wolf *et al*. 2007). Unlike the Urban Water Balance model there is less focus on required meteorological data with only daily precipitation and potential evapotranspiration values needed. Aquacycle contains options to apply water management techniques to the UWB and was evaluated using data from Woden Valley, Canberra, Australia (Mitchell *et al.* 2003). UVQ was utilised to investigate the UWB and transport of contaminants for a number of cities around the world as part of the AISUWRS research project (Wolf *et al*. 2007).

2.13.2 Urban Parameterisation Schemes used in Meso-scale Models

The second type of model identified were urban parameterisation schemes used in global and mesoscale numerical weather models. The parameterisation schemes identified in Table 2.14 are the Urban Hydrological Element model, UHE (Berthier *et al*. 2004, 2006), the urbanised Submesoscale Soil Model, SM2-U (Dupont *et al*. 2006) and the combined Town Energy Balance and Interaction Soil-Biosphere-Atmosphere scheme, TEB-ISBA (Lemonsu *et al*. 2007). Each scheme differs in complexity and focus but in essence are formed of a number of surface and subsurface layers with the aim of modelling the surface water balance using inputs from a numerical model (typically net radiation and precipitation) and generating output for use in the next model time step (in this case using the UWB the evapotranspiration). All the schemes presented have mixed land cover (urban and natural surface types) each of which have individual surface and hydrologic properties weighted by their relative areal coverage of a particular grid box. Unlike the dedicated UWB models these parameterisations focus only on the external water system.

The UHE model (Berthier *et al*. 2004, 2006) is a water budget model which simulates storm water runoff and soil infiltration. It is formed of two main layers a surface layer with three possible land cover types (natural, paved and building roof) and a soil layer (formed of an upper and lower sublayer for infiltration purposes) which takes the form of a fine mesh grid. In addition to these layers there is a storm water drainage system which is represented as a trench collecting all available runoff as well as seepage from soil water (this acts in both directions depending on soil moisture conditions). The original version of the model (Berthier *et al*. 2004) considered evapotranspiration (and infiltration) by applying a 'mixed' boundary condition on modelled soil moisture, observed rainfall and potential evapotranspiration. This model was further developed to include an dual evapotranspiration scheme based on the Penman-Monteith-Rutter-Shuttleworth equation (Grimmond & Oke 1991) for paved and roof surfaces and a scheme based on Feddes *et al.* (1988) to calculate the potential evaporation and transpiration for the natural surface types (Berthier *et al*. 2006). The model was then evaluated with data from the Rezé field site using site specific parameters and observed meteorological and hydrological variables.

SM2-U is a surface parameterisation scheme (Dupont et al. 2006) for a mesoscale model with four levels (lower atmosphere, surface layer, root zone and deep soil) and a rudimentary drainage network as used in Berthier *et al*. (2004). It is an extension of Noilhan and Planton's (1989) ISBA scheme, with the addition of four urban surface types each with their own surface properties (Dupont *et al.* 2006): building roofs, paved surfaces, vegetation over paved surfaces and paved surfaces under vegetation. These extra surface types resulted in modification to a number of terms related to the water balance. The scheme was evaluated with data from three measurement sites two rural and the suburban site at Rezé which also allowed comparison of the runoff running through the drainage network with the original version the UHE (Berthier *et al.* 2004). It was concluded that the SM2-U scheme performed well annually and in summer storm events in comparison to UHE but was poor at simulating winter storms due to moisture infiltration to and from the drainage network not being modelled. Berthier *et al*. (2006) showed large differences (37%) in modelled evapotranspiration (see Section 2.7) due to differences in methodology, which has implications on the comparability of urban water budgets using the two models.

TEB-ISBA scheme was used by Lemonsu *et al*. (2007). The scheme has three layers (surface and two soil layers) and four surface types (three ISBA vegetation surfaces: bare soil, soil between vegetation and vegetation; and the TEB urban canyon). An off-line simulation of the TEB-ISBA water balance was undertaken using meteorological data to force the model and input parameters from the literature relevant to a suburban area. Comparison with data from the Rezé catchment, resulted in improved parameterisation of surface infiltration through roads due to discrepancies between modelled and observed runoff (Lemonsu *et al.* 2007).

2.13.3 Hydrology Models

The third type are hydrology models; for example, Semi-Urbanised Runoff Flow, SURF (Rodriguez *et al.* 2000), the Water and Energy transfer Processes (WEP) model (Jia et al. 2001), the Urban-Runoff Branching Structure MOdel, URBS-MO (Rodriguez *et al.* 2008) and the Urban FORest Effects-Hydrology (UFORE-Hydro) model (Wang *et al.* 2008). These are typically composed of two parts: a surface scheme and a hydrological flow model (described as a natural, anthropogenic or a combination drainage system) that moves water through the study catchment. These models are typically used for studying city wide sewer and drain

performance during rainfall events (Rodriguez *et al*. 2008), assessing impacts of runoff pollution (Rodriguez *et al*. 2000), the simulation of the effects of urban areas on natural catchment flows and flooding (Jia *et al*. 2001) and the impact of urban trees and canopy interception on runoff and evapotranspiration (Wang *et al.* 2008).

Table 2.14: Summary of urban water balance and urban/ semi-urban hydrological models that utilise the UWB framework to varying degrees (by year of publication).

Developer	Name	Location/Scale/ Timestep	Process	Methods
Grimmond *et al*. (1986)	Urban Water Balance (UWB)	Vancouver, Canada/ Pr, Nh, Ci/ Hourly-Daily	Urban water balance	- Empirical relations, water use surveys & meteorological observation.
Thielen & Creutin (1997)	No name given	St. Denis, Paris, France/ Ci,Rg/ hourly	Combined atmospheric and urban hydrology	Module based. Clark model (storm model) & CAREDAS (Hydrology module)
Rodriguez *et al.* (2000)	Semi-Urbanized Runoff Flows (SURF)	Rezé, Nantes, France/ Nh, Ci/ event based	Urban runoff	GIS/DEM land use model. Simplified water budget (r=P-E-infiltration)
Jia *et al*. (2001)	Water and Energy transfer process model (WEP)	Ebi River, Japan/ Pc, Rg/ hourly	Surface water and energy model	Combined Infiltration & Runoff model Surface energy balance
Mitchell *et al*. (2001)	Aquacycle	Woden Valley, Canberra, Australia/ Pr, Nh, Ci/daily	Urban water balance	Empirical relations. Water use profile data.
Berthier *et al.* (2004)	Urban Hydrological Element (UHE)	Rezé, Nantes, France/ Nh,Ci/ model dependent	Urban hydrology model (runoff, E and soil)	Penman-Monteith-Rutter-Shuttleworth method. Fine grid soil process model. surface reservoir method for runoff.
Diaper & Mitchell (2006)	Urban Volume and Quality Model (UVQ)	Australia/Nh,Cw/daily	Urban water balance	Empirical store, flow and simple process model
Dupont *et al.* (2006)	Sub-mesoscale Soil Model, urbanized version (SM2-U)	Rezé, Nantes, France/ Nh, Ci, Rg/ ?	Urban parameterisation scheme.	Extension of ISBA. E = specific humidity gradient. Runoff from surface reservoir method
Lemonsu *et al.* (2007)	Town Energy Balance – Interactions between Soil, Biosphere & Atmosphere (TEB-ISBA).	Rezé, Nantes, France/ Nh, Ci/ ?	Urban soil-vegetation-atmosphere interaction.	Multi surface parameterisation scheme. -Focus on surface infiltration.
Rodriguez *et al.* (2008)	Urban Runoff Branching Structure MOdel (URBS-MO)	Rezé & Gohards, Nantes, France/ Nh, Ci/ unknown	Urban Hydrology	Surface model based on UHE. Hydological flow/drainage model
Wang *et al*. (2008)	UFORE – Hydro (Urban FORest Effects – Hydrology model)	Dead Run, Baltimore, USA/ Pc/ hourly	Urban Hydrology	Tree interception model (LAI, % cover) E, Penman-Monteith TOPMODEL (infiltration & runoff)

To assess the current state of modelling of the UWB a comparison is required of available models with the premise of assessing their performance in modelling urban flows and stores, identifying key parameters (and optimal magnitudes) and determining the required level of complexity. Once evaluation is complete an UWB modelling framework can be established so that future models and tools can be built that are focused on accurately modelling the UWB for assessing the effectiveness of sustainable water management techniques and strategies as well as allowing improvements to numerical model surface parameterisations.

2.14 Conclusion – The Future

It is apparent that there is potential to improve modelling and measurement of the UWB. Some key requirements to take the study of the UWB forward are:

- Improvements are required in the study of the spatial variability of precipitation over urban catchments (e.g. through the use of radar, satellite data and rain gauge networks).

- There is little knowledge on the magnitude and processes associated with anthropogenic water vapour. A mixture of field and modelling studies are required to address this.
- Work is required to improve the measurement and modelling of urban evapotranspiration. There is a need for long term measurements over urban areas with results made available through dedicated database or communicated clearly in the literature.
- A greater number of reliable measurements of atmospheric and hydrological variables that are representative of urban areas are required for use in determining UWB. This includes improvements to the determination of surface temperature, precipitation and surface/subsurface moisture and the magnitude of pipe leakage.
- Process monitoring and modelling studies are required to improve the parameterisation and simulation of processes within the UWB (e.g. infiltration, the role of vegetation in urban areas and pipe leakage).
- An UWB monitoring/observation framework is required to ensure the optimal number of variables are observed using methods that are scientifically sound while ensuring that observed values are representative of urban areas. In addition to this an UWB reporting framework for presenting results from UWB studies is required, so that future work can learn from the limitations, assumptions and methods used in previous studies while observations and parameters can be compared in confidence.
- An UWB model comparison study is required with the aim of developing a modelling framework from which future models can be based upon. The study needs to address the current state of modelling with particular focus made on processes, parameters and complexity. The framework needs to consider the ability of models to assess sustainable urban water management techniques and investigate the effects of climate change.
- Accurate databases (topography, land uses and land covers, vegetation and soil properties, urban morphology such as roof slopes and road conditions, sewer networks, surfaces connected to network, etc.) are absolutely critical for interpretation of observations and modelling.

Part III: Carbon

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3.1 Introduction

Given urban populations are continuously increasing, the study of urban CO_2 and pollution fluxes becomes very crucial, both to guaranty a pleasant and healthy environment for urban dwellers and to ensure that the effects of urbanization do not have harmful repercussions on large scale climates.

In the context of enhanced global warming, cities affect greenhouse gas sources and sinks both directly and indirectly, but the link between urbanization and global climate change is complex (Sánchez-Rodríguez et al., 2005). The urbanization process produces radical changes in the nature of the surface and atmospheric properties of a region (as noted in previous sections). Radiative, thermal, moisture and aerodynamic characteristics are transformed and the natural solar and hydrological balances are changed.

Odum and Barrett (2005) classified urban areas as techno-ecosystems, having entirely new arrangements compared to natural ecosystems. Urban communities consume material and energy inputs, processing them into usable forms, and eliminate the wastes as outputs of the process. In addition, within an urban system, it is important to consider the urban sprawl and the urban area's footprint; the latter is the area of land needed to provide the necessary resources and absorb the wastes generated by a community. It incorporates water and energy use, uses of land for infrastructure and different forms of agriculture, forests, and all other forms of energy and material "inputs" that people require in their day-to-day lives. It also accounts for the land area required for waste assimilation.

Urban areas are recognized to constitute the major sources of the CO_2 emitted into the atmosphere. Svirejeva-Hopkins et al. (2004) suggest that more than 90% of anthropogenic carbon emissions are generated in cities. In particular, humans activity (human respiration, domestic heating, and airplane) and automobile produce more than 80% input of CO_2 into the urban environment (Koerner and Klopatek, 2002). For this reason, the knowledge of urban carbon exchanges is important to better understand the interaction of natural and anthropogenic processes that control the role of cities in carbon budgets (Walsh et al., 2004). Much attention has been also focused on the increase of air pollutants (Gratani et al., 2000; Brack, 2002; Moreno et al., 2003) and therefore on air quality.

In this context, the BRIDGE project, using the conceptual framework of urban metabolism, will focus on the interrelation between energy and material flows and urban structure. The objective is to provide the means to quantitatively estimate the various components of the urban metabolism from the local to the regional scales, and to assess the environmental impacts of the above components.

In this discussion, an overview relative to the carbon cycle and pollutant issues in the urban environment is presented. A brief description of the current state of measurement and modelling urban studies of both CO_2 and pollutants will be given. The inventories methodology for the estimation of carbon and pollutant emissions will be presented as a useful tool for model parameterization and validation.

3.2 Atmospheric chemical compounds

A brief outline of the main chemical compounds of the atmosphere is here reported. They are constituents that pose a risk to health, which may alter visibility, and have a high impact on human activities or the environment. Currently, one of the major points of debate in international issues is the greenhouse effect and related climate change (Tiezzi, 2002; Keller, 2003). The Kyoto Protocol agreement aims to reduce collective greenhouse gas (GHG) emissions by at least 5% compared to 1990 levels for the period 2008-2012. Important links have been established between regional air pollution and climate change, although these are currently hardly considered in policy-making (e.g. RIVM, EFTEC et al., 2001; Syri et al., 2001; Mayerhofer et al., 2002; Van Harmelen et al., 2002). First, some substances directly influence both climate change and regional air pollution, for instance,

sulphur dioxide (SO_2) and nitrogen oxides (NOx). Second, the emissions of greenhouse gases and regional air pollutants originate to a large extent from the same activity, i.e. fossil fuel consumption.

The two major groups of chemical species in the atmosphere are:

1. <u>Greenhouse gases</u>

- ***Oxides of carbon***. CO_2 is an essential element for life, therefore it is not considered a pollutant. It derives from the complete combustion of fuel in the presence of oxygen. Globally averaged surface atmospheric CO_2 concentrations are 379 ppm (IPCC, 2007) and are estimated to increase by 50–100 ppm by 2100 (Friedlingstein et al., 2006). In urban areas, CO_2 concentrations are higher varying between 370 ppm to more than 500 ppm depending on the measurements site (residential area, suburban, city center, etc.) and on the city dimension (e.g. metropolitan area) and population density. The presence of the domestic heating also affects CO_2 concentration resulting in increasing it (Maltese et al., 2009). Carbon monoxide (CO) is mainly generated by the incomplete combustion of carbonaceous materials. Anthropogenic sources play the major role in the CO emission through the processes of internal combustion engine. In urban area and highways CO has the maximum concentration. Other sources include metal processing industries, gasoline refineries, and paper processing factories. However, the most important source of this compound is the cigarette smoke. In high concentrations CO is potentially lethal. It is a relatively stable gas and it can be removed from the atmosphere by oxidation to carbon dioxide (CO_2), or absorption by the ocean and soil.
- ***Hydrocarbon.*** Most of these compounds (Hc) originate from the natural decomposition of vegetation. Anthropogenic release is very small, but it is important these compounds are potentially harmful to vegetation and humans. They are also highly reactive and facilitate the formation of photochemical smog. Hc mainly derives from fossil fuel combustion and from evaporation from gasoline. So, vehicles are the primary source and traffic density is the main factors that control their emission. In Nagoya, one of the typical urban areas of Japan (Aikawa et al., 1995b), authors found that the annual mean concentration of methane increased in three years from 1.94 ppm to 1.98 ppm.
- ***Oxides of nitrogen.*** Natural organic decomposition from soil and oceans originates nitrogen emission. Anthropogenic sources are represented by combustion of fuel under pressure and heat resulting in the fixation of nitrogen and oxygen to form nitric oxide (NO), a harmless gas. In the atmosphere, this gas rapidly oxidizes to nitrogen dioxide (N_2O), an irritant gas. Vehicles, coal and natural gas burning, fertilizer and explosives factories are the principal sources of nitrogen oxides.

2. <u>Air pollutants</u>

- ***Particulate Matter* (PM).** Atmospheric particle derive both from natural sources (about 90%) and human activities. Anthropogenic sources are mainly associated with combustion, industrial processing, and surface disturbance due to building activities.
  PM is a very complex pollutant, not only because particles typically consist of a mixture of substances, but also because some of the substances that make up the particles are semi-volatile. Semi-volatile substances can exist in the air both as particles and vapours (i.e. gases). The mass of semi-volatile PM (e.g. ammonium nitrate and some organic compounds) is not static but can instead change frequently as the substances respond to the changing meteorological, physical and chemical conditions that they encounter while moving through the air.
  Particles are either liquid or solid and their size vary from greater than 100 µm to less than 1 µm. Dust, grit, fly ash and visible smoke (greater than 10 µm) tend to settle out relatively rapidly after emission and create problems mainly close to the source. Fine particles (less than 1 µm in diameter) remain in the boundary layer for several days and can create a smoky haze in weak ventilation condition. The smallest particles, ultrafine particles (diameter < 0.1 µm) are of particular interest due to their importance in health related effects.
  Particles are mainly composed by carbon and silica, but also iron, lead, manganese, cadmium, chromium, copper, nickel, beryllium and asbestos. PM is emitted directly to the air (primary PM), and it also forms in the air (secondary PM) from precursor gases such SO_2, NO_x, VOC and NH_3. Sources of primary PM include soot (elemental carbon, or EC) emitted directly from combustion of fossil fuels; metals such as iron, lead, mercury and cadmium; elements of soil and road dust; bio-aerosols (i.e. particles containing or composed of living micro-organisms such as fungal spores and mould); and salt (e.g. road salt and oceanic sea-salt).

Ambient levels of particles can be elevated year-round, and in urban areas the levels are typically higher in the mornings and evenings, reflecting traffic patterns. Particles can travel very large distances and affect areas thousands of kilometres away from the sources of the emissions.

- ***Sulphur compounds***. Main sulphur compounds present in the atmosphere are: sulphur dioxide (SO_2), hydrogen sulphide (H_2S), sulphurous (H_2SO_3), sulphuric acid (H_2SO_4), and sulphate salts. About two-third of all atmospheric sulphur comes from natural sources and H_2S represent the maximum amount deriving from bacterial action. Anthropogenic activities (combustion of sulphur-bearing fuels, oil refineries, and ore smelters) mainly release SO_2. Oil refineries also emit H_2S.
- ***Minor and secondary pollutants***. Minor pollutant means that these compounds are emitted in small quantities or are restricted to small areas, but they can be significant for some aspects (e.g. toxicity) and should not be neglected. Such compounds are: hydrogen fluoride (from fertilizer factories), toluene (from paint solvents), radioactive substances, and ammonia. Secondary pollutants include a wide range of substances created by chemical reactions between two or more pollutants, or between pollutants and natural atmospheric constituents. They are, for example, ozone (O_3), peroxyacetyl nitrates (PAN) and aldehydes.

US and Europe established thresholds of tolerance for these compounds to safeguard human health. The U.S. Environmental Protection Agency (EPA) provides the Air Quality Index (AQI) with a standardized scale from 0 to 500 for all criteria pollutants (EPA, 1999) (Table 3.1). Also the European Union has developed an extensive body of legislation which establishes health based standards and objectives for a number of pollutants in air. These standards and objectives are summarised in the table below (Table 3.2). These apply over differing periods of time because the observed health impacts associated with the various pollutants occur over different exposure times.

Table 3.1: AQI values, categories, and pollutant concentration thresholds for the criteria pollutants. EPA (1999).

AQI	AQI category	O_3 (ppb) 8 h[a]	O_3 (ppb) 1 h	$PM_{2.5}$ (μg m^{-3})	PM_{10} (μg m^{-3})	CO (ppm)	SO_2 (ppm)	NO_2 (ppm)
0–50	Good	0–64	—	0.0–15.4	0–54	0.0–4.4	0.000–0.034	([b])
51–100	Moderate	65–84	—	15.5–40.4	55–154	4.5–9.4	0.035–0.144	([b])
101–150	Unhealthy for sensitive groups	085–104	125–164	40.5–65.4	155–254	9.5–12.4	0.145–0.224	([b])
151–200	Unhealthy	105–124	165–204	65.5–150.4	255–354	12.5–15.4	0.225–0.304	([b])
201–300	Very unhealthy	125–374	205–404	150.5–250.4	355–424	15.5–30.4	0.305–0.604	0.65–1.24
301–400	Hazardous	([c])	405–504	250.5–350.4	425–504	30.5–40.4	0.605–0.804	1.25–1.64
401–500	Hazardous	([c])	505–604	350.5–500.4	505–604	40.5–50.4	0.805–1.004	1.65–2.04

[a] Areas are generally required to report the AQI based on 8-h ozone values. However, there are a small number of areas where an AQI based on 1-h ozone values would be more precautionary. In these cases, in addition to calculating the 8-h ozone index value, the 1-h ozone index value may be calculated and the maximum of the two values is reported.

[b] NO_2 has no short-term NAAQS and can generate an AQI only above a value of 200.

[c] When 8-h O_3 concentrations exceed 374 ppb, AQI values of 301 or higher must be calculated with 1-h O_3 concentrations.

Improving the air quality studies is important to provide benefits for public health, planning business and activities, responding to emergency situations, planning controlled burns, aiding fire-fighters in setting up command posts, managing or fighting fires, and protecting them-selves from exposure to smoke.

Two subgroups in air quality forecasts exist: health-alert and emergency-response prediction. The first focuses on ozone, particulate matter, nitrogen dioxide, carbon monoxide, sulphur dioxide, and lead. The second focuses on situations where chemical, biological, or nuclear materials are unexpectedly emitted into the atmosphere (accident, natural hazards or terrorist attacks) and where the source is poorly described.

Air quality prediction includes the depiction of the present chemical state of the atmosphere in the urban zone and on the regional (meso-β) scale (Dabbert et al., 2004): the dynamic nature of the Planetary Boundary Layer (Stull, 1988) influences the concentration and residence time of pollutants in the atmosphere and, hence, air quality.

Table 3.2: Pollutant concentration thresholds from the new Directive 2008/50 EC of 21 May 2008 on ambient air quality and cleaner air for Europe.

Pollutant	Concentration	Averaging period	Legal nature	Permitted exceedences each year
Fine articles (PM2.5)	25 µg m^{-3}***	1 year	T10 D5	n/a
Sulphur dioxide (SO_2)	350 µg m^{-3}	1 hour	D5	24
	125 µg m^{-3}	24 hours	D5	3
Nitrogen dioxide (NO_2)	200 µg m^{-3}	1 hour	T10	T10
	40 µg m^{-3}	1 year	T10*	n/a
PM10	50 µg m^{-3}	24 hours	D5**	35
	40 µg m^{-3}	1 year	D5**	n/a
Lead (Pb)	0.5 µg m^{-3}	1 year	D5 (or 1.1.2010 in the immediate vicinity of specific, notified industrial sources; and a 1.0 µg/m3 limit value applies from 1.1.2005 to 31.12.2009)	n/a
Carbon monoxide (CO)	10 mg m^{-3}	Max.daily 8 h mean	D5	n/a
Benzene	5 µg m^{-3}	1 year	D10**	n/a
Ozone	120 µg m^{-3}	Max. daily 8 h mean	T10	25 days averaged over 3 years
Arsenic (As)	6 ng m^{-3}	1 year	T12	n/a
Cadmium (Cd)	5 ng m^{-3}	1 year	T12	n/a
Nickel (Ni)	20 ng m^{-3}	1 year	T12	n/a
Polycyclic Aromatic Hydrocarbons	1 ng m^{-3} (as conc. of Benzo(a)pyrene)	1 year	T12	n/a

* Under the new Directive the member State can apply for an extension of up to five years (i.e. maximum up to 2015) in a specific zone. Request is subject to assessment by the Commission. In such cases within the time extension period the limit value applies at the level of the limit value + maximum margin of tolerance (48 µg/m3 for annual NO2 limit value). ** Under the new Directive the Member State can apply for an extension until three years after the date of entry into force of the new Directive (i.e. May 20011) in a specific zone. Request is subject to assessment by the Commission. In such cases within the time extension period the limit value applies at the level of the limit value + maximum margin of tolerance (35 days at 75µg/m3 for daily PM10 limit value, 48 µg/m3 for annual PM10 limit value). *** Standard introduced by the new Directive.
D5 Limit value entered into force 1.1.2015; T10 Target value enters into force 1.1.2010 T12 Target value enters into force 1.1.2012 D10Limit value enters into force 1.1.2010

Although cities have been shown to minimize stability effects, pollutant dispersion measurements show that plume spread is dependent on stability (McElroy, 1997). As the boundary layer becomes less stable, mixing is suppressed less and the emission velocity of pollutant increases. In the unstable regimen, emission velocity decreases with increasing instability. Aerosol fluxes also show that the downward transport is increased during unstable conditions. It is not clear whether this is a meso-scale feature due to the upstream air advected into and above the city, or if this is micro-scale feature resulting from different rates of heating (and cooling) of urban surfaces. In fact, direct measurement of fluxes indicates how emission of particles from the urban canopy is strongly dependent upon the thermal emission from the city and thermal stability of the atmosphere (Nemitz et al., 2000; Dorsey et al., 2002; Mårtensson et al., 2006; Schmidt and Klemm, 2008). Largest emissions occur during the day with higher surface heating.

3.3 The urban carbon budget

The global carbon cycle in urban environment can be represented as in Figure. 3.1. Input into a biological system must pass through and the amount of waste is dependent on the amount of resources required. A balance sheet of inputs and outputs exists. Natural and anthropogenic inputs participate to storage carbon in four main

pools (plants, soil, building and human products, people and animals). Urban carbon outputs consist then in respiration by live creatures and soils, burning of fossil fuel and waste decomposition. One can manage the waste produced, but energy is required to turn it into anything useful. All carbon products will end up as CO_2 and it is not possible to recycle them any further without enormous energy inputs that in themselves have associated wastes. This is an entropy factor in urban metabolism.

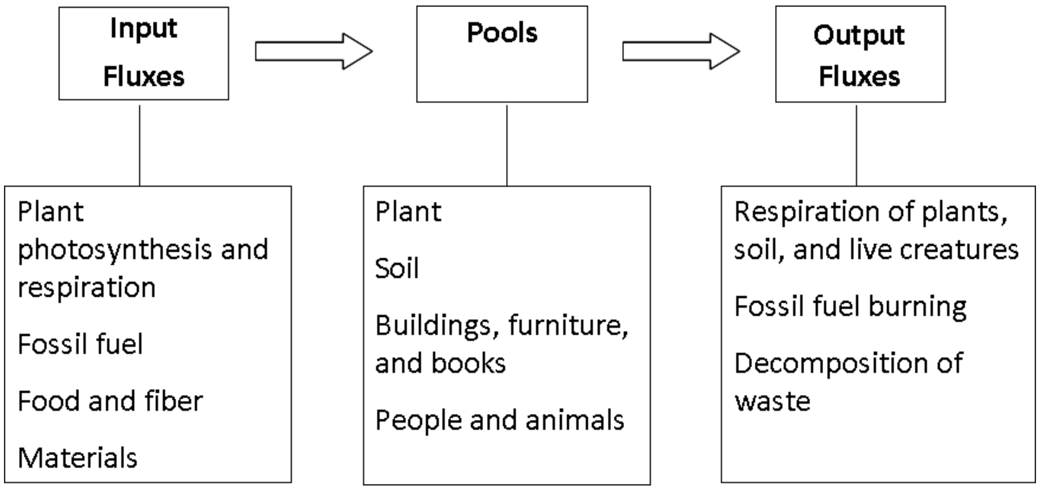

Figure 3.1: Carbon input, pools, and output in urban ecosystem (modified from Churkina, 2008).

The urban ecosystem is a heterotrophic system which cannot support metabolism by internal production alone. It requires imports of materials and energy produced in external ecosystems and store or transform to waste products that have significant effects on downwind or downstream ecosystems. So, large areas of natural and semi-natural ecosystems are needed to maintain urban demands and are fueled by energy mostly produced by fossil fuel burning (Churkina, 2008): most of the carbon and energy used by a city comes from outside the city boundaries or from urban footprint. These exchanges can be seen as "metabolism" of industry, commerce, municipal operations, and households. The direction of these exchanges is shown in Figure 3.2 where also the key elements of carbon budget in an urban system (Figure. 3.2) are represented as follow:

(1) *Driving forces and urban matrix*, referred to people and climate. In effect, the location of an urban system is mainly defined by people’s priorities (e.g. fertile land, proximity to the trade roads, esthetical value of a location), and climate (e.g. people prefer to live in warmer climates) (Churkina, 2008). The spatial and structural patterns of urban areas are considered by the urban matrix. Its description includes extents and properties of impervious areas and vegetation growing in cities as well as their management practices.

(2) *Vertical fluxes of carbon*, which are divided in fluxes of natural -ecosystem photosynthesis and respiration- and anthropogenic - fossil fuel burning, decomposition of waste, and human breath.

(3) *Horizontal fluxes of carbon*, which are mostly driven by human activities, include transfers of food and fiber from natural ecosystems into urban systems and flows of trash from the area of urban sprawl into landfills located usually in the urban footprint

Inputs of both vertical and horizontal fluxes are distinguished from a point of view of matter that is gained (Decker et al., 2000), both actively (through human work) and passively. *Active inputs*, including materials that are stored or transformed, become part of the urban built environment. *Transformed inputs*, food and fuel, providing mass and/or energy needed by humans or their tools to function, must be supplied constantly (Kinzig and Socolow, 1994; Matson et al., 1997). *Passive inputs*, as air, consist in build-up of gas phase carbon dioxide (IPCC 1996; US Census Bureau, 1999).

With regard to the **outputs**, material flow associated with the air exiting metropolitan areas can be usefully conceived of as a superposition on the background tropospheric input (Decker et al., 2000). The typical urban area injects most of the background tropospheric species and adds to the concentration distribution as well (Finlayson and Pitts, 1986; Seinfeld and Pandis, 1998).

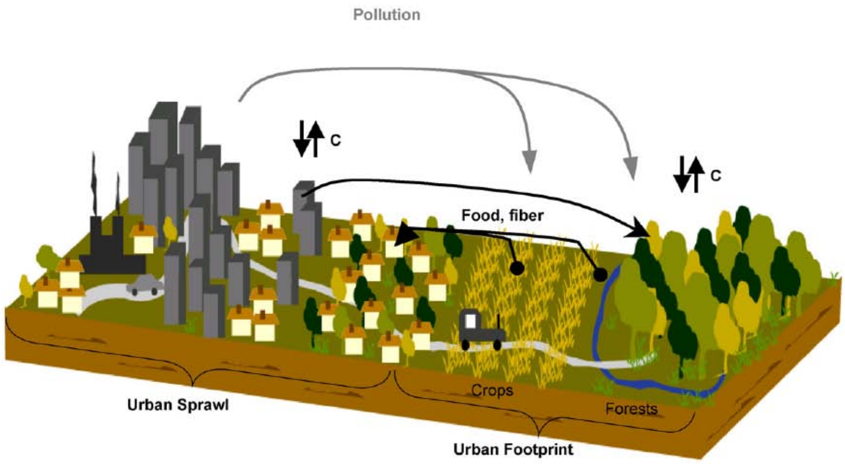

Figure 3.2: Urban system. The vertical and horizontal carbon fluxes are shown with black arrows (from Churkina 2008).

The two anthropogenic activities that have the major impacts on the global carbon cycle in urban environment are fuel combustion and changes in land use and land cover (Pataki et al., 2006b). The direct effect of fossil fuel emissions is arguably the most significant effect of urbanization on the carbon cycle. In the 1990s, fossil fuel combustion resulted in emission of 6.3×10^3 Mt yr^{-1} of carbon globally, of which 3.2×10^3 Mt yr^{-1} was retained in the atmosphere (IPCC, 2001). About 40% of total fossil fuel emissions in the US are attributed to the transportation and residential sectors (WRI 2005).

Conversion of agricultural or other land to urban and exurban uses may also modify the carbon uptake and carbon storage. Zhao et al. (2007) found that low-density exurban development can register more gross primary production (GPP) than agricultural land, due to the large portion of vegetation replacing crops. In addition, soils in urban parks and lawns can store an amount of carbon more than double that one stored in native grassland or agricultural fields (Kaye et al., 2005; Golubiewski, 2006). Higher rates of carbon uptake and storage can also derive from an increase in trees cover on land converted into urban and exurban areas. Results also showed that the net effect of urban land-use conversion will depend partly on the characteristics of the native or rural ecosystem replaced. For arid regions, the net overall effect of urbanization may be higher productivity rates, with the potential to actually increase net C storage; in more humid environments, the net effect may be a reduction in soil C storage (Pouyat et al., 2002; Golubiewski, 2003). A recent study of Churkina et al. (2009) showed that the total carbon storage in the US urban and exurban areas was 18.5 Pg in 2000, highlighting that human settlements can store more carbon than the US croplands, which store 14 ± 7 Pg C (King et al., 2007). Contribution of soil in carbon storage was predominant (64%), and then 20% of the carbon was stored in vegetation, 11% in landfills, and 5% in buildings.

The form and structure of cities, in addition to building characteristics, may well influence the amount of CO_2 emitted and stored. In addition, demographic trends interact with urban forms in ways that have an impact on the emissions of CO_2. The pattern of urban development may be the key to determining the amount of CO_2 emissions and the ability of cities to reduce them (Alberti, 2008). The number and size of households affect the number and size of housing and associated energy uses. Furthermore, the spatial distribution of residential and commercial housing units affects commuting patterns and transportation choices with important consequences for fossil fuel consumption. Future strategies should then consider these aspects to reduce carbon emissions and to increase carbon storage in human settlements.

Urbanisation processes have also an impact on the amount of the atmospheric pollution. In fact, pollution levels increase in urban areas and there is the need of long-term research to understand the ecosystem dynamics (Gratani et al., 2000). Pollutants, when released into the atmosphere, pose a direct and serious hazard to living organisms in general, and to humans in particular (Pearson et al., 2000). As seen before, the effects of urban areas on the environment extend much further than the city's boundaries, so that biogeochemistry of significantly larger areas is affected (Churkina, 2008). Air pollutants originating in a city and transported outside of its limits can adversely affect regional climate and atmospheric chemistry (Rosenfeld, 2000; Crutzen, 2004) as well as the vegetation in that region.

In urban areas, air concentrations of pollutants are controlled by the balance between those factors which lead to pollutant accumulation and those which lead to pollutant dispersal. Emission and dispersion processes are influenced by a wide range of temporal and spatial scale (Figure 3.3). The nature of emission and the state of the atmosphere determine the amount of pollution at a site. The amount and type of pollutant are controlled by several factors as the rate of emission and the physical and chemical nature of the pollutants, the shape of the emission area, the duration of the releases and the effective height of pollutants injection (Oke, 1987). Dispersion of pollutant after release is then controlled by atmospheric motion as stability conditions, wind and turbulence. The pollutants may be also transformed by physical and chemical processes due to water vapour or droplets present in the atmosphere, air temperature, solar radiation intensity, and other atmospheric substances. Pollutants are removed from the atmosphere by precipitation-related processes (*scavenging*), by gravitational settling, or by surface adsorption and impaction.

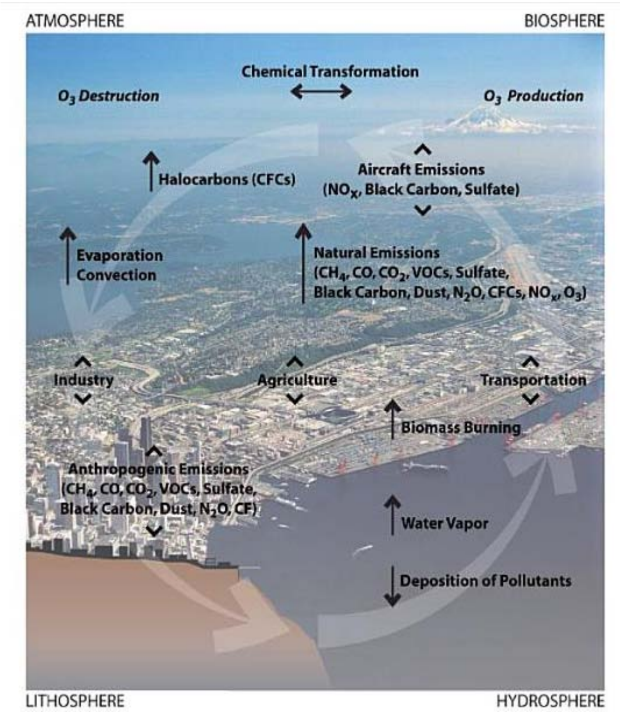

Figure 3.3: The atmospheric cycle in the urban landscape (from Alberti, 2008).

3.4 Carbon and pollutant measurements

Studies of carbon and pollutant differ depending on the spatial scale. In the micro-scale (10^1–10^2 m) studies, spatial differences in processes occur in response to variability in building/canyon dimensions and orientations. GHGs grow until they are constrained laterally by the buildings (e.g., Dabberdt and Hoydysh, 1991). The station have to be set in order to achieve climate observations free of extraneous microclimate signals to characterize local climate.

In local scale (10^2–10^4 m) GHGs studies, we can consider the **blockscale** or **neighborhood-scale** (100–1000 m) regime. It deals with GHGs that grow laterally to encompass several buildings and may grow vertically to the top of the buildings. Here it is important to account for the wind and turbulence within the urban canopy, but also to account for the effects of the buildings themselves, for example, the heat from their walls in the daytime (Grimmond and Oke 1991; Arnfield and Grimmond 1998; Grimmond and Oke 1999), or the way GHG stalls in their wakes (e.g., Dabberdt et al. 1994). **Meso-scale** (10^4–10^5 m) studies are instead referred to the city in its entirety, differentiated from its surroundings, areas of forest, agriculture, etc.

From micro- to meso-scale, several methods are used to measure carbon and pollutant concentration and estimate related fluxes. In this section a brief description of the main methods at local and regional scale are reported. Experimental results deriving from urban carbon and pollutant studies are also presented.

3.4.1 Methods

Local and regional carbon balance may be estimated from atmospheric measurements using eddy covariance (Grimmond et al., 2002, 2004a, b; Nemitz et al., 2002; Soegaard & Moller-Jensen, 2003) or with measurements of CO_2 concentration using different methodologies.

Chemical methods have evolved since 1812 for measuring the CO_2 content of air (gravimetric, titrimetric, volumetric or manometric) at local scale. The Pettenkofer titrimetric method – being simple, fast and well understood - was used as the optimal standard method for more than 100 years after 1857 (Pettenkofer, 1858; Pettenkofer, 1862; Pettenkofer and Voit, 1866; Kauko and Mantere, 1935). Different scientists calibrated their methods against each other, and by sampling gas with known CO_2 content. Most studies between 1812 and 1961 were obtained from rural areas or periphery of towns. Beck (2007) analysed data on CO_2 levels over a 150 years period determined by classical chemical techniques from station located throughout the northern hemisphere showing these historical methods were reliable and produced good quality results.

A **laser photoacoustic spectrometry** (Sigrist, 1994) was used for experimental CO_2 and trace gas measurements both in rural and urban environment. A mobile system was used in Zurich (winter 1985-86), in Biel and Dubendorf near Zurich (summer 1986) to monitor concentration of CO_2, ethene (C_2H_4), and ozone.

The **isotopic composition of CO_2** is a powerful tool for identifying sources and sinks in the regions strongly affected by anthropogenic activity, distinguishing emissions from fossil fuel combustion and natural sources (Kuc and Zimnoch, 1998; Widory and Javoy, 2003; Pataki et al., 2003, 2005b). Fossil fuels contain no radiocarbon by virtue of their age in contrast to respiration of modern carbon from plants and soils (Zondervan & Meijer, 1996; Meijer et al., 1997; Takahashi et al., 2001, 2002). The oxygen isotope composition of CO_2 is also generally distinct for respiration respect to the combustion-derived CO_2, depending on the isotopic composition of local water plant available water (Florkowski et al., 1998; Pataki et al., 2003, 2005a). In addition, it is possible to distinguish different fuel types such as gasoline or natural gas, which have distinct carbon isotope ratios. In fact, for anthropogenic sources, CO_2 derived from natural gas combustion is more isotopically depleted in stable ^{13}C than CO_2 derived from gasoline combustion (Tans, 1981; Andres et al., 2000; Pataki et al., 2005a). As a result, natural gas combustion, gasoline combustion, and biogenic respiration have distinct combinations of isotope tracers that can be used to solve for the proportional contributions of each source to total CO_2 in the atmosphere. Isotope-tracer technology was used by Pataki et al. (2003) in a year-long study in *Salt Lake City*, Utah. Authors found that biogenic respiration contributed up to 60% of local, non-background, CO_2 during the vegetation growing season. In contrast, natural gas combustion constituted 30–70% of local CO_2 in the wintertime depending on ambient temperatures and time of day, with colder temperatures resulting in increased natural gas consumption from residential furnaces, and evening rush-hour periods showing a greater contribution from vehicular traffic (Pataki et al., 2003, 2005b, 2006). In *Krakow*, measurements were performed in the period 1994-2000 (Kuc et al., 2003), and the carbon isotope composition of CH_4 in the urban atmosphere (natural gas used in the city) allowed to recognize in leakages of this gas in the distribution network as the main anthropogenic source of CH_4 in the local atmosphere. CO_2 and CH_4 fluxes from urban atmosphere resulted higher than in regional background.

Different types of **gas analyzers** were used in several studies, but the method based on infrared ray (IR) absorption has been found to possess the advantages of wide measuring range, fast response, high sensitivity, and good selectivity. Therefore, most of studies use a high speed, high precision, non-dispersive infrared gas analyzer that accurately measures densities of carbon dioxide and water vapour in turbulent air structures. CO_2 gas exhibits very strong absorption bands in the 2.7 µm and 4.26 µm regions of the infrared spectrum making it possible to analyze CO_2 gas concentration with very high resolution. At 4.26 µm, no other atmospheric gas, apart from CO_2, displays strong absorption characteristics. This wavelength is then appropriate for analyzing CO_2 gas concentration.

Infrared gas analyzers were used to monitor CO_2 concentration over different cities around the world. Results showed that CO_2 concentration depends on the measurement site location, with higher concentration in the city

center or suburban areas than in rural areas (Day et al., 2000; 2002; Idso et al., 2001; George et al., 2007), and on the season and time of the day, with higher values in winter than in summer and during the night (Woodwell et al., 1973; Aikawa et al., 1995a; Reid and Steyn, 1997).

Micrometeorological techniques are actually fully used to measure the net exchange in the lower part of the atmosphere. This approach has been initially employed for several ecosystems, as grasslands, forests and wetlands, for example in FLUXNET program (Baldocchi et al., 2001a,b). Based on such measurements, important data are emerging on the role of these different ecosystems, spatial and temporal (daily, seasonal, and annual) variability and controls. *Eddy-Covariance* (EC) technique is the most used micrometeorological method to measure fluxes at local scale. Theoretical bases for the EC method was initially established by Sir Osborne Reynolds (Reynolds, 1895).

Urban surface affects atmospheric fluxes, so the application of EC technique to urban areas represents a challenge for making representative measurements of source and sink, and for generalizing results to larger scale. Each local scale surface type (e.g. distinct neighbourhood) generates an internal boundary layer that grows with fetch at a rate depending on the roughness and stability. Differently from rural conditions where a height:fetch ratio varying from 1:10 (unstable conditions) to 1:500 (in stable conditions) (Garratt, 1992; Wieringa, 1993) is required, urban environment needs a ratio of 1:100 because of neutral conditions generated by thermal and mechanical turbulence associated with the heat island and large roughness.

As described in the Part 1 (Figure 1.1), within the urban canopy layer we can recognize two sublayers: the roughness sublayer (RSL) and the internal boundary layer (ISL). In order to account for the urban features affecting GHGs and pollutant measurements, sensors' exposure should be above the RSL, but below the ISL of the area of interest. In fact, within the RSL, entities are not well mixed and carbon dioxide concentrations are likely to be highly variable whereas the ISL represents the layer with constant flux and turbulent fluxes are expected to be spatially homogeneous throughout the layer. Therefore, instruments have to be mounted at a height at least twice the mean height of the roughness elements to ensure that measurements represent the integrated response at the local-scale (Grimmond and Oke, 1999; Kastner-Klein et al., 2000; Rotach, 2000), and in fairly uniform (both in terms of surface cover and roughness elements) neighbourhoods (Figure 3.4).

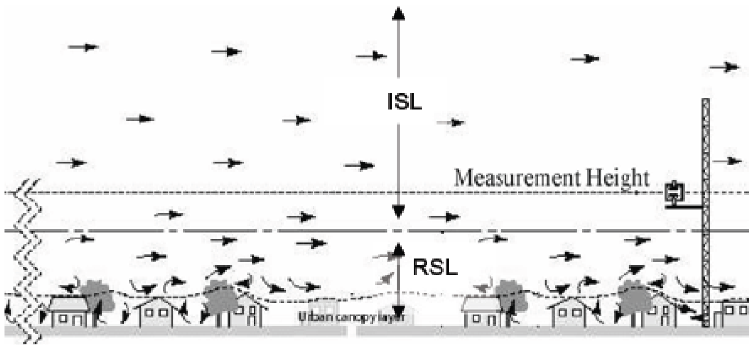

Figure 3.4: Ideal measurements height over an urban environment (Masson et al., 2002, cited by Walsh et al., 2004).

EC measurements can be made by a fixed tower or by instruments for measuring air velocity, pressure, temperature, and fluxes mounted at the front of the aircraft. Regional surface fluxes and concentrations can be obtained by flying at several levels and linearly extrapolating the flux profile to the surface, since for conserved species in well-mixed boundary layer the flux varies linearly with height (Guthrie and Veblen, 1992). Nakazawa et al. (1997) collected air samples using aircraft measurements in the troposphere over Russia in the summers of 1992, 1993, and 1994. They analyzed vertical profiles of CO_2, CH_4, N_2O, and CO. Results showed that CO_2 concentration increased with height over all locations. CO_2 concentrations also increased moving from wetland to tundra and the city of Moscow. Isotopic analysis showed that CO_2 releases were primarily caused by human activities such as fossil fuel combustion. In contrast to CO_2, CH_4 concentration decreased with height and higher values were found over wetland. N_2O concentrations were quite constant over all locations, while CO showed a small gradient over natural wetlands, taiga, and tundra. Over the urban area of Moscow, high values of CH_4, CO_2 and CO concentrations were observed due to emissions of the respective gases by human activities. This method requires high costs for aircraft equipment, so few studies on gas exchanges in urban areas exist.

EC measurements are also used in combination with **optical particle counters** to estimate pollutant fluxes (Longley et al., 2004). Such an optical particle counter measures the optical diameter of each particle. Then, the conversion to geometric size depends upon the shape and refractive index of the particle, which is not only generally unknown, but which will also vary from one particle to the next, and will vary with measurement location (in terms of micro-environment and geographically) and with time. Sampling in an environment dominated by materials of atypical refractive index, such as elemental carbon, some error may be introduced into the sizing. Urban aerosols are generally heterogeneous due to local anthropogenic sources and the presence of particles with short residence times, and hence these uncertainties may be larger. EC flux system was also used to measure the aerosol number flux particles (Mårtensson et al., 2006; Järvi et al., 2009b).

The **disjunct eddy covariance** (DEC) method can be applied for the determination of size-resolved turbulent vertical particle fluxes directly by applying an electrical low pressure impactor (ELPI) for the size-resolved particle concentrations. The advantage of the DEC method is the possibility to obtain turbulent fluxes when using slower instruments by allowing increased measurement intervals between two samples, but keeping the sampling duration itself short enough to capture turbulent fluctuations. The scalar (the size-resolved particle concentrations) is measured with lower sample frequencies as limited by the response time of the employed measurement devices. Fluxes obtained with DEC methodology and with a conventional EC method showed a good agreement, so the fluxes determined with the DEC are a very good approximation of the fluxes obtained with the direct EC (Schmidt and Klemm, 2008).

Another instrument used to measure fine and ultrafine aerosol concentrations is the dual **Differential Mobility Particle Sizer** (DMPS). It allows to estimate the total number and size segregated, and to determine the size segregated chemical speciation of the aerosol (Williams et al., 2000; Ketzel et al., 2003; Järvi et al., 2009).

3.4.2 Experimental results

CO_2 concentration

Near-surface CO_2 concentrations have been documented in several cities across the world (Vancouver, Canada; Kuwait City, Kuwait; Mexico City, Mexico; Basel, Switzerland; Nottingham, UK; Phoenix, USA) to evaluate the dynamics of atmospheric CO_2 over time (Berry and Colls, 1990a,b; Reid and Steyn, 1997; Idso et al., 2001; Nasrallah et al., 2003; Velasco et al., 2005; Vogt et al., 2006).

Most of studies have been performed over European cities than in other regions. A list of studies measuring CO_2 concentration in urban environments with summary details of measurements and key results is given by Grimmond et al. (2002). Table 3.3 gives an overview of the recent studies on CO_2 concentration over urban cities in the world. These studies have documented that CO_2 concentrations are greater in urban environments (reaching 500 ppm in some cases) than natural ecosystem due to anthropogenic sources (Kuc, 1991; Nakazawa et al., 1997; Kuc and Zimnoch, 1998; Kuc et al., 2003).

The majority of studies analyzed daily and diurnal fluctuations in CO_2 concentrations and concluded that the major source of CO_2 is from vehicular traffic as peak CO_2 concentrations correlate to high traffic volume during workdays and is significantly reduced at weekends (Berry and Colls, 1990a; Aikawa et al., 1995a; Idso et al., 1998, 2001, 2002; Hom et al., 2003; Nasrallah et al., 2003; Velasco et al., 2005). Reid and Steyn (1997) compared concentrations measured over a busy suburb of Vancouver, Canada using an infrared gas analyzer to those modelled using a numerical multiple-box transport and mixing model. The diurnal cycle is divided into 4 stages: (1) early morning peak; (2) draw-down from morning peak to afternoon minima; (3) afternoon minima, and (4) the rise from lower afternoon concentrations to high nocturnal CO_2 concentrations. Higher average enhancements at night were about 18 ppm than during the day (9 ppm) (Reid and Steyn, 1997). Similar diurnal cycle in CO_2 concentration was found in Basel, Switzerland (Vogt et al., 2003; 2006) and Chicago, Illinois (Grimmond et al., 2002). In the latter city, the mean CO_2 concentration for 13 days was 384 ppm, and the early morning peak was attributable both to anthropogenic (largely traffic), biospheric (nocturnal respiration), and meteorological factors (Grimmond et al., 2002). A highest peak of 650 ppm (76% higher than the low of 369 ppm) was found over a two week period in Phoenix, Arizona (Idso et al., 2001). A longer study, showed values ranged from a daily minimum of 390 ppm rising to a daily maximum of 491 ppm, although a maximum value of 619 ppm was attained (about 60% of the lower value) (Idso et al., 2002). Annual value of CO_2 concentration

strongly correlated to the traffic volume were also found in Rome, Italy (Gratani and Varone, 2005), showing an increase of 30% in CO_2 concentration from 1995 (367 ppm, mean yearly value) to 2004 (477 ppm, mean yearly value), and in Kuwait City (Nasrallah et al., 2003) with a mean CO_2 concentration value of 369 ppm.

Dense urban areas with large populations and high vehicular traffic showed significantly different microclimates compared to outlying suburban and rural areas. A transect from Baltimore, Maryland, city centre (urban site), to the outer suburbs of Baltimore (suburban site) and out to an organic farm (rural site) (George et al., 2007). The highest concentration on average across the 5 years of the study was at the urban site (488 ppm), the lowest at the rural site (422 ppm), and the suburban site intermediate to the other two sites (442 ppm). Smaller differences were observed in Nottingham, UK (5 ppm between a rural location and an urban city site,Berry and Colls, 1990a), and 19 ppm and 12 ppm was found in Phoenix (Day et al., 2000; 2002) and Vancouver (Reid and Steyn, 1997), respectively.

Local weather conditions are very important for atmospheric CO_2 content. Woodwell et al. (1973), using an infrared gas analyzer for 159 days in New York, found a difference in CO_2 concentration in winter and summer of 19 ppm. Similar results were found in Nagoya, Japan (Aikawa et al., 1995a) showing lower values in summer and higher in winter both for CO_2 and CH_4 concentration, related to general circulation of the atmosphere, and some characteristic temporal changes due to atmospheric stability and emission sources. At local scale, CO_2 fluctuations were also greatest closer to the ground and diminished with height (Woodwell et al., 1973).

CO_2 fluxes

The earlier observations of carbon fluxes using EC method over urban areas were short campaigns. Only recently long-term measurements have been conducted within urban areas. Christen et al. (2006) reported 5 urban long-term monitoring sites (Baltimore, Basel, Berlin, Tokyo, and Rome) equipped with EC system for CO_2 flux measurements, and compared the results. In the Table 3.4 are summarized the main urban studies on carbon fluxes.

In urban areas, CO_2 fluxes are almost always positive; so, the urban surface is a net source of CO_2 (Grimmond et al., 2002; Hom et al., 2003; Vogt et al., 2003; Grimmond et al., 2004b; Moriwaki and Kanda, 2004; Velasco et al., 2005; Coutts et al., 2007; Crawford et al., 2009). In *Edinburgh*, UK, CO_2 fluxes ranged from -12 to 135 µmol m^{-2} s^{-1}, with an average of 22 µmol m^{-2} s^{-1} (Nemitz et al., 2002). From September to December, in *Florence* (Italy), daily CO_2 flux was always a positive term corresponding on average to 25.8 µmol m^{-2} s^{-1}, but NEE varied from 12.7 µmol m^{-2} s^{-1} up to 35.9 µmol m^{-2} s^{-1} when domestic heating was turned on (Maltese et al., 2009). The magnitude of CO_2 flux in *Marseille* was approximately twice the magnitude of the average values reported in a more residential, heavily treed neighbourhood of Chicago (Grimmond et al., 2002). Again in Marseille, Salmond et al. (2005) tested the hypothesis that CO_2 concentrations can be used as a tracer to identify characteristics of venting of pollutants and heat from street canyons into the above-roof nocturnal urban boundary layer using the novel analytical technique. An investigation on the *Tokyo* bay revealed that the sea breeze affects fluxes (the bay was a net sink of carbon) and CO_2 concentration raised to an annual average of 500 ppm (Oda et al., 2006). The highest downward fluxes (-8 µmol m^{-2} s^{-1}) when compared to other urban studies was found in *Helsinki*, Finland (Vesala et al., 2008).

The study in *Baltimore* represented the first permanent flux tower (about 4 months) to measure carbon flux in an urban/suburban environment (Hom et al., 2003). The clear relationship of carbon flux with traffic was highlighted by a study conducted in the city center of *Rome* (Miglietta et al., 2004). It revealed a relevant decrease in carbon emissions during the days with traffic limitations imposed to private cars. The clear diurnal pattern (highest emissions during the morning and the lowest ones in the nighttime) was found for CO_2 flux, as for concentration, (Nemitz et al., 2002; Velasco et al., 2005) and VOCs (Velasco et al., 2006).

The role of vegetation in CO_2 emission mitigation is not so clear. Studies conducted in *Vancouver* (Walsh et al., 2004), *Mexico City* (Velasco et al., 2005), and *Melbourne*, Australia (Coutts et al., 2007) revealed that urban vegetation is not able to offset CO_2 emissions by respiration and fuel combustion, but in *Salt Lake Valley* differences in behaviour in CO_2 sequestration between residential and rural areas was found. The vegetative cover at the residential area had significantly effects on the diurnal variation in CO_2 fluxes showing a carbon uptake in the midday, in contrast with the rural area that was a net source of CO_2 (Ramamurty and Pardyjak, 2007). Also an area at 10 km south of *Denver's*, Colorado, urban center, surrounded by commercial and

residential areas, was a small source of CO_2 during the vegetation growing season or a net sink under certain conditions (Anderson and Taggart, 2002).

GIS analysis has turned out a useful tool to investigate how different sources of CO_2 (anthropogenic sources, physical geography, land-use types, and meteorology of the area) contribute to the CO_2 dome (Koerner and Klopatek, 2002). Results showed that soil CO_2 emissions do not have a large contribution to the CO_2 dome (15.8% of the total CO_2 emission for *Phoenix*, AZ), while anthropogenic sources appear to be largely responsible for it.

Remote sensing technique has revealed a means to continuously measure the CO_2 budget, too. It was combined with EC measurements in a **multi scale** approach over the city of *Copenhagen*, Denmark (Soegaard and Moller-Jensen, 2003). By this approach, authors were able to separate the contribution of the terms of the carbon balance: the mobile sources (traffic); the static local sources (heating and service/household); the semi-static sinks (carbon assimilation through photosynthetic activity; and the remote sources (power plants). A separated section will be dedicated to remote sensing techniques used in urban studies (Section 3.7).

Pollutant

Several studies have been conducted to estimate trend and concentration of pollutants over the major cities (Table 3.5). Some studies estimated emissions through direct measurements (using eddy covariance method or particle sizers system) and other studies developed some models to simulate and evaluate the climate changes and air pollution concentrations.

First direct measurements of size-segregated particle fluxes over a city were performed in *Edinburgh* in 1999 (Nemitz et al., 2000), using fast response eddy-correlation systems (EC). Nemitz et al. (2000) described the emission fluxes of fine and coarse particles on a diurnal and seasonal basis, whereas Dorsey et al. (2000; 2002) reported on the methodology and results for total particle number fluxes. In *Chicago* (Lestari et al., 2003) PM showed three peaks: a fine peak (0.43 µm) and two coarse particulate peaks (6.75 and 28.58 µm). Sulfate and nitrate existed in both the fine (< 2.5 µm) and coarse (> 2.5 µm) particles. Dry deposition fluxes of sulfate and nitrate were between 1.0-4.0 and 0.5-3.9 mg (m^{-2} day^{-1}), respectively, and 99% of the nitrate and sulfate deposition was due to the coarse particles modes.

Measurements of traffic-related particle number concentration are reported in tunnel studies (Kirchstetter et al., 1999; Kristensson et al., 2003; He et al., 2008) near highways, and in urban street or background locations (Allen et al., 2001; Jamriska and Morawska, 2001; Ruuskanen et al., 2001; Wehner et al., 2002). An example of values of pollutant concentration in urban area is given by the study in the metropolitan area of *Karachi*, Pakistan (Ghauri et al., 1994). CO_2 concentration was found to exceed 370 ppm in the busy urban streets, and CO was around 9-10 ppm. Main sources of HCs emissions were from vehicles and careless handling of petrol and allied products. Combustion of petrol/diesel of vehicles also produced high level of NOx compounds. The ozone concentration was higher in the central areas (40-50 ppb) of the city than in the coastal area (25 ppb). Daily total suspended particulates (TSP) were monitored in the 13 sites and the five sources of heavy metals were identified: soil-limestone (for Al, Ca, Mn, Fe, Co, Hf, Th), sea spray (for Na, Cl), fossil fuels (for Se and non-marine sulphates), vehicular traffic (for Br, Pb), and metal plating/air conditioning (for Br). High levels of Pb were found (3-7 fold higher than average). Pronounced diurnal cycles in the particle number fluxes well correlated with traffic activity, were found in several studies (Longley et al., 2004; Mårtensson et al., 2006; Schmidt and Klemm, 2008; Järvi et al., 2009a, b).

Studies revealed that the importance of traffic decreases with increasing particle size (Järvi et al., 2009a). Ketzel et al. (2003) found the maximum traffic effect at particle sizes of 20–30 nm. Turbulent mixing is also an important variable in determination of the particle levels and it has an opposite effect on different sized of particles. With increasing turbulence, particles are mixed into a larger air volume and their concentrations decreases on ground level. At the same time, the re-suspension of larger particles increases.

Size distribution is strongly affected by wind direction. In particular, studies showed that the wind influences the emissions of particles of minor size (Nemitz et al., 2000; Williams et al., 2000; Longley et al. 2003), demonstrating that wind-driven re-suspension is the main mechanism for coarse aerosols from the city centre.

Emissions distribution and amount differ due to the city area investigated. In 1999, a study investigated the pollutants composition on the road and under the tunnel system of the Underground of *London* (Sitzmann et al., 1999). On the road, carbon particles were clearly the dominant particle type in the road traffic samples. The size distribution of the aerosol was typical of vehicle emissions. Samples from the Underground showed a distinctly different size distribution and element composition (higher concentration of Fe/Si-rich particles). The particle mass concentration in the Underground was 10 times higher than those at the roadside due to the numbers of particulate sources, restricted ventilation and significant particle re-suspension during train movements.

3.5 Inventories

It is known that emissions inventory is useful tools to monitor the atmospheric composition in urban area, identify main sources of GHGs emissions and to locate where these gases are emitted. Inventory also constitutes a tool for measured data validation and model parameterization and development. Therefore, attempts to quantify the role of urban areas on the global carbon budget and pollutant release have focused largely on **inventories of emissions** (from estimates of fossil fuel consumption, cement production, control of air pollution, etc) and sinks (the amount of carbon sequestered in urban vegetation based on biomass estimates) (Jo and McPherson, 1995; Johnson and Gerhold, 2003).

Advantages of emission inventories are: (1) the *ability to highlight the predominant source contributions and the main opportunities for emission control;* (2) *their use to assess the implementation of control strategies.* With a particular regard to urban emissions inventory, Ball and Radcliffe (1979) identified five main applications:

- An emission map, showing the geographic distribution of emissions, can be an important aid in land use planning by identifying parts of the region that are likely to be subject to high levels of pollution, and the location of pollution sources in relation to sensitive areas.
- An emission inventory can help in estimating the cost of introducing controls, and identifying who should bear those costs.
- Emissions inventories can help to design efficiently a monitoring network. The design of monitoring networks is important, if meaningful data are to be obtained. The data may be required to assess the exposure of the population to a particular pollutant, or to demonstrate compliance (or non-compliance) with air quality standards. An emission inventory will indicate, for example, where the highest concentrations of pollution are likely to be found, or which areas are the most representative. It can thus help to ensure that monitoring equipment is appropriately located.
- Ideally, it should be possible to use an emission inventory, in conjunction with an atmospheric dispersion model, to predict short term pollutant concentrations at ground level during forecast adverse weather conditions. When these techniques are fully developed, they could be used to alert authorities to possible air pollution incidents and to determine strategies for avoiding them.
- An emission inventory can be used, alone or in conjunction with an atmospheric dispersion model, to assess trends in air quality. By altering the inputs to the emissions inventory in a way that simulates future conditions (such as a change in fuel use), it is possible to make prediction about the impact on air quality.

Different approaches are used to estimate carbon and pollutant emissions (Puliafito, 2006). In particular, the top-down approach and the bottom-up approach are generally proposed to estimate the emissions from urban areas (Table 3.6). The selection of one of these methods will depend on the availability of input data and the desired spatial and temporal resolution.

Table 3.3: Examples of studies measuring CO_2 concentration in urban environments, by scale from smaller to larger.

Location	Site measurements	Scale	Observation Period	Purpose	Variables and Techniques	Averaged CO_2 concentration	Reference
Phoenix , Arizona	Center and the metropolitan edge	Micro-scale	14 March-3 April 2000; 14 days, Jan. 2000	Temporal patterns of CO_2 concentration contrasting vegetation types ; to study the CO_2 dome	Infrared gas analyzer	398 ppm in the center and 384 ppm at the metropolitan edge; 500 ppm in the center	Day et al., 2000; 2002 Idso et al., 2001
Paris, France	Various locations suburbs, country	Street level	Several days 1997 to 1999	To investigate different sources of atmospheric CO_2 in Paris	Isotope characterization of air samples	Higher CO_2 conc.. laboratories and classrooms than streets and gardens	Widory and Javoy, 2003
New York , US	Brookhaven, Long Island	Local scale	159 days from 1965 to 1971	To estimate CO_2 exchange	Infra-red gas analyser	19 ppm difference from winter to summer; highest conc.. > 500 ppm	Woodwell et al. ,1973
Nottingham, UK	City center and transect rural to urban areas	Local scale	8 months	To investigate on CO_2 and SO_2 concentration changes along a transect from rural to urban areas	Infra-red gas analyzers	5 ppm higher in urban area	Berry and Colls, 1990a,b
Biel, Swizterland	Suburb	Local scale		To dev elope photoacoustic systems for trace gas monitoring	Laser photoacoustic spectrometry	335 ppm	Sigrist, 1994
Nagoya, Japan	Nagoya Univ. campus	Local scale	3 years (1991-1993)	To continuously monitoring CO_2 concentration in a urban area	Infrared absorption spectrometer	381 ppm in 1991, 382 in 1992, 377 in 1993	Aikawa et al., 1995a
Vancouver, Canada	Two suburban neighborhoods	Local scale	June 1993	To describe the diurnal CO_2 variation of atmosphere in a coastal city	Infrared gas analyzer. Developed a numerical multiple-box tansport and mixing model	375 ppm	Reid and Steyn, 1997
Krakow, Poland	City center	Local scale	1983-1994	To analyse changes in CO_2 concentration and carbon isotope composition	Molecular sieve for sampling	373 ppm in 12 years;	Kuc and Zimnoch, 1998
Phoenix, Arizona	Residential area	Local scale	315 days in 2000	To study the CO_2 dome	Infrared gas analyzer	390 ppm	Idso et al., 2002
Kuwait City, Kuwait	Suburban site	Local scale	1996-2003	To compare CO_2 concentration in Kuwait City with Phoenix study	CO_2 gas analyser	369 ppm	Nasrallah et al., 2003
Salt Lake City, Utah		Local scale	One year	To monitor CO_2 sources and anthropogenic and biogenic effects on CO_2 seasonal cycle	Tunable diode laser absorption spectrometer (TDL) to measure CO_2 mixing ratio and isotope composition		Pataki et al., 2003; 2006
Rome , Italy	City center	Local scale	1995-2004	To evaluate CO_2 concentration in relation to traffic volume	CO_2 gas analyser	From 367 ppm to 477 ppm in the central zone in 9 years	Gratani and Varone, 2005
Baltimore, Maryland	Highly vegetated area	Local scale	2002-2006	To characterize carbon fluxes and carbon dioxide concentrations from a highly vegetated residential area	CO_2 gas analyzer	488 ppm as averaged of 5 years	George et al., 2007
Krakow, Poland	City center	Regional scale	1983-1988	To investigate CO_2 concentration and its isotopic composition in Southern Poland	Molecular sieve for sampling and multi annual trend analysis	In 1983 CO_2 conc. was about 27-30 ppm higher than Mauna Loa	Kuc, 1991

Table 3.4: Examples of studies measuring CO_2 fluxes in urban environments, by scale from smaller to larger.

Location	**Site measurement**	**Scale**	**Observation Period**	**Purpose**	**Variables & Techniques**	**Averaged CO_2 concentration**	**Reference**
Basel, Switzerland	Dense urban city center	Street level and local scale	Summer 2002	To investigate fluxes and profiles of CO_2	EC method	420 ppm as a maximum; daytime values are in the range of 10 to 20 $\mu mol\ m^{-2}\ s^{-1}$	Vogt e al., 2003; 2006
Edinburgh, UK	City centre	Local scale	28 October-30 November 2000	To estimate the urban heat budget and CO_2 emissions	EC method	373 ppm; average flux was 22 $\mu mol\ m^{-2}\ s^{-1}$	Nemitz et al., 2002
Baltimore, Maryland	Highly vegetated area	Local scale	May-September 2001	to characterize the carbon fluxes and carbon dioxide concentrations from a highly vegetated residential area	EC method	386 ppm in contrast to 511 ppm in the city center	Hom et al., 2003
Marseille, France	City center	Local scale	June-July 2001	To estimate energy, mass, and momentum fluxes	EC method		Grimmond et al., 2004b
Rome, Italy	City center	Local scale	February-June 2004	To investigate CO_2 flux in Rome	EC method	45 g CO_2 m^{-2} day^{-1} during the coldest period and 22 g CO_2 m^{-2} day^{-1} in the early summer.	Miglietta et al., 2004
Tokyo, Japan	Kugahara residential area Tokyo bay	Local scale	May 2001-April 2002 December 2004-August 2005	To investigate seasonal and diurnal energy and mass fluxes	EC method	398 ppm; annual emission of 3352 gC m^{-2} 500 ppm	Moriwaki and Kanda, 2004 Oda et al., 2006
Vancouver, Canada	Suburban area	Local scale	August 2001-January 2003 July-August 2008	To provide long-term CO_2 flux measurements Local scale CO_2 flux measurements from two contrasting suburban sites and a rural reference station	EC method		Walsh et al., 2004 Crawford et al., 2009
Mexico City, Mexico	Densely populated section	Local scale	April 2003	To measure CO_2 flux and VOCs fluxes	EC method	388 ppm ; average flux of 0.41 mg m^{-2} s^{-1}	Velasco et al., 2005; 2006
Melbourne, Australia	Two residential sites	Local scale	February 2004-June 2005	To quantify the magnitude of CO_2 flux and spatial variability in a Southern Hemisphere city	EC method	365 ppm; average CO_2 flux of 21 gCO_2 m^{-2} d^{-1} at Preston site, higher at Surrey Hills	Coutts et al., 2007
Salt Lake Valley, Utah	Two suburban sites	Local scale	Summer 2005	To compare CO_2 fluxes at two urbanized sites	EC method	396 ppm residential area; higher concentration in the pre-urban area	Ramamurty and Pardyjak 2007
Florence, Italy	City center	Local scale	September-December 2005	To evaluate EC method for urban area and make CO_2 sources partitioning	EC method	388 ppm without housing heating, 424 ppm with housing heating on; Av. CO_2 flux of 25.8 $\mu mol\ m^{-2}\ s^{-1}$	Maltese et al., 2009
Copenhagen, Denmark	City centre	Multi scale approach	1 year	To estimate the carbon balance	EC method, remote sens.	35 g CO_2 m^{-2} day^{-1}	Soegaard and Moller-Jensen, 2003

Table 3.5: List of air quality measurement studies, by scale

Location	Scale	Observation Period	Purpose	Techniques	Pollutant values	Reference
CONCENTRATIONS						
Copenaghen, Denmark	Street and roof-levels	May - November 2001	To separate street and roof level emissions	Two Differential Mobility Particle Sizers (DMPS) systems	No_x conc.: 42.7 ppb at street level and 11.3 ppb on the roof CO conc.: 0.70 ppm at street level and 0.25 ppm on the roof PM10 conc.: 21.4 µm/m^3 at street level	Ketzel et al., 2003
Karachi, Pakistan	Local scale	15 days in May 1990	To assess air quality in Pakistan	Not given	CO_2 > 370 ppm; CO between 9 to 10 ppm; O_3 higher in city center (40-50 ppb) than in coastal area (25 ppb); daily mean TSP of 250 µg m^{-3}; Pb was 3-7 fold higher than average	Ghauri et al., 1994
Nagoya, Japan	Local scale	3 years (1991-1993)	Continuously monitor CH_4 concentration	Gas chromatograph with a flame ionization detector (GC/ FID)	1.94 ppm in 1991, 1.93ppm in 1992, 1.98 ppm in 1993	Aikawa et al., 1995b
London, UK	Local scale	1 week	To assess the personal exposure of cyclists and Underground train users to PM5 and characterize these particles.	Computer-Controlled Scanning Electron Microscopy (CCSEM) and energy dispersive X-ray detection (EDX) to analyse samples.	Road traffic: carbon particles are the main particle type (66%) Underground: Fe/Si-rich particles are 53% of total; PM conc. London Underground – 10x higher than measured in traffic generated aerosol	Sitzmann et al., 1999
Edinburgh, UK	Local scale	October 1999	To determine the fine and ultra fine aerosol within the city	Dual Differential Mobility Particle Sizer (DMPS) to monitor the size distribution of aerosol		Williams et al., 2000
FLUXES						
Manchester, UK	Street canyon level	2 weeks in October 2001	Diurnal cycle of ultrafine particle concentrations in a busy street canyon	EC system; two optical particle counters (PMS ASASP-X)	Number of fine particle fluxes (0.1 µm < Dp< 0.5 µm) ranged from 3700 cm^{-2} s^{-1} to 1100 cm^{-2} s^{-1};	Longley et al., 2003; 2004
Edinburgh, UK	Local scale	October 1999; Two 3-weeks campaigns in May & Oct. 1999	Direct measurements of total aerosol fluxes	EC system and a condensation particle counter (CPC)	Average aerosol number fluxes in the range 11 nm <Dp< 3 µm varied between 9000 and 90000 cm^{-2} s^{-1}	Dorsey et al. 2000; 2002; Nemitz et al., 2000
Chicago. Illinois	Local scale	Spring, Summer , and Fall of 1994 and 1995	To investigate on the size distribution and dry deposition of PM, sulfate and nitrate	Wide Range Aerosol Classifier; smooth surface deposition plate; deposition model	Sulfate conc .on varied from 1.0-7.0 (fine) to 0.3-1.0 (coarse) µg/m^3 Nitrate conc. 0.3-5.3 µg/m^3 in fine and 0.3-2.9 µg/m^3 in the coarse fraction Sulfate dry deposition flux = 1.0-4.0 mg/(m^2 day); nitrate dry deposition flux = 0.5-3.9 mg/(m^2 day)	Lestari et al., 2003
Stockholm, Sweden	Local scale	49 days from 19 March to 6 May 2002	Quantify, parameterise aerosol source number flux F (particles m^{-2} s^{-1})	EC method	Aerosol particle number concentration was 70845 cm^3 at the street level and 10070 cm^3 at rooftop	Martensson et al., 2006
Munster, Germany	Local scale	Summer 2007	Direct determination of highly size-resolved turbulent particle fluxes	Disjunct Eddy Covariance method (DEC) and a 12-stage electrical low pressure impactor	Average diurnal aerosol number flux varied 0 - 1.6 x 10^7 m^{-2} s^{-1} (Di=0.04 to 2.0 µm)	Schmidt and Klemm, 2008

Table 3.6: Urban approaches for emissions estimates (modified from Puliafito, 2006)

TOP-DOWN	**BOTTOM-UP**
population data	intracity detailed studies
consumption surveys, source-destiny census	in situ measurements
emission factors, fuel consumption, etc - useful for annual mean values and long term trends	short termed variations
	small spatial scales

The **top-down approach** is normally used to calculate emissions from fixed sources (i. e., residential, industrial, commercial, etc.), assuming a relative low temporal or seasonal variability based on ambient temperatures, school activity, holiday's distributions, and so on. The inventory computes yearly energy consumption, using population density, economic activity and emission factors. Additionally to the emissions from fuel combustion, it is necessary to collect the industrial emissions from the production process, together with their proper engineer information, i.e. stack height, diameter, exit fluxes and emission rates, which will feed the input data requirements of a dispersion model calculation (Puliafito and Allende, 2007).

The **bottom-up approach** is, by contrary, used to measure in situ fluxes characterized by short term variations and small spatial scales. For example, to estimate the emissions from road vehicles in a bottom-up approach, it is necessary to record traffic fluxes and speed in several streets. The bottom-up method has high temporal resolution (normally an hourly base) with variable spatial resolution. It is clear, then, that this approach is more accurate but requires high data density. The emissions in each street are calculated using emission factors based on average traveled distances for each vehicle category.

At the moment, there are a number of national and international guidelines for preparing emission inventories on a more or less nation-wide level (Table 3.7). For **greenhouse gases** we have the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 1995). This reference, a guideline for the development of emission inventories at a national level, uses a top-down methodology (Loibl et al., 1993; Orthofer and Winiwarter, 1998). It uses available information for greater emission areas and breaks down the overall emissions into sub-units using data for source strength and emission generating activities, or, in the absence of this information, source specific statistical data. However, the application of this approach at the urban scale is not always appropriate, and, as seen before, a bottom-up methodology often being advisable. This second approach divides the study area into grid cells (or administrative units), for each of which all emission sources and activities are accounted for.

Some examples for **classical air pollutants** (SO_2, NOx, etc.) are, for instance, the CORINAIR methodology (Bouscaren et al., 1992) and the EMEP/CORINAIR `Atmospheric Emission Inventory Guide book (McInnes, 1996). These original works led to other versions such as the US-EPA emission inventory guidelines (US-EPA, 1977, 1980, 1995, 1999). Another effort is the EUREKA Environmental Project EUROTRAC I Subproject GENEMIS (Ebel, 1997). One of the main tasks of this project was to put emphasis on the spatial and temporal resolution of emissions in order to provide data for macro-scale dispersion modeling. At present, the Subproject GENEMIS is continuing in the context of the new Project EUROTRAC II. Sometimes it is necessary to obtain an emission inventory with a high temporal and spatial resolution, and in these cases a special methodology is developed (Costa and Baldasano, 1996; Baldasano, 1998).

Table 3.7: List of the main emission inventories on a nation-wide level.

Inventory	Reference	Period	Purpose	Variables
OECD/MAP Project	OEDC 1990	1983 - 199	To assess pollution by large scale photochemical oxidant episodes in Western Europe and evaluate the impact of vari emission control strategies for such episodes..	sulphur dioxide - SO_2 nitrogen oxides - NOx, and vol organic compounds - VOC, including natural emissions.
CEC Environment Directorate (DGXI)	CITEPA, 1988	1980 - 198	To collect data on emissions from all relevant sources (amo them domestic heating and transportation) in order to produ database for using in the study of air pollution problems an base policy measures in the field of air pollution control.	SO_2, NOx, VOC and particulat
CORINAIR 1990	CORINAIR 19	1990 - 199	produced a more developed nomenclature (source sector sp SNAP90 - involving 11 main sectors (among them public power, cogeneration and district heating plants; commercia institutional and residential combustion plants) extended the list of pollutants; To provide a complete, consistent and transparent air pollut emission inventory for Europe in 1990 within a reasonable scale to enable widespread use of the inventory for policy, research and other purposes.	sulphur dioxide (SO_2) oxides o nitrogen (NOx) non-methane volatile organic compounds (NMVOC) ammonia carbon monoxide methane nitrous oxid carbon dioxide
Project EUROTRAC I Subproject GENEMIS		1997-2002	to improve methods, models and emission factors for the generation of emission data for atmospheric models and the assessment of the accuracy of emission data	
IPCC Guidelines for National Greenhouse G Inventories	IPCC 1995			CO_2, CH_4 and N_2O; HFCs, PF and SF_6; NOx, CO and NMVO
Emission Database for Global Atmospheric Research (EDGAR) 2.0	PBL, Bilthoven (NL) and JRC-I Ispra (IT)	1990	To provide global annual emissions for 1990 of greenhouse gases and precursor gases, both per region and on a 1x1 gri all anthropogenic sources. Similar inventories were compile for a number of CFCs, halons and methyl bromide, methyl chloroform.	greenhouse gases CO_2, CH_4, N precursor gases CO, NOx, NMVOC and SO_2
Emission Database for Global Atmospheric Research (EDGAR) 3.2	Olivier et al., 2001a,b; Olivie 2002).	1990-1995	EDGAR 2.0 has been updated to EDGAR 3.2: an update an extension from 1990 to 1995 for all gases and extended tim series for direct greenhouse gases to 1970-1995; and inclus of 1970-1995 emissions of the new 'Kyoto' greenhouse gas aimed at analyzing the long-term changes in the atmospheri budget of trace gases and aerosols; exploiting (often under-utilised) existing data sets from ground based stations, aircr and satellite instruments, integrating these into common datasets; developing tools for the analysis, interpretation an exploitation of the data ; formulating recommendations for future measurement strategies makes it possible to estimate a given energy- and agricultural scenario, the costs and environmental effects of user-specified emission control policies estimates emission reduction potentials and costs f range of greenhouse gases and air pollutants and quantifies resulting impacts on air quality and total greenhouse gas emissions considering the physical and economic interactio between different control measures.	direct greenhouse gases CO_2, C and N_2O; HFCs, PFCs, SF6
Inventory	**Reference**	**Period**	**Purpose**	**Variables**
REanalysis of the TROpospheric chemica composition over the p 40 years		1960 -200	To create global gridded data sets for anthropogenic and vegetation fire emissions of several trace gases to provide a reasonable estimate of emissions including their seasonal a interannual variability in the major burning regions of the	gas-phase species and anthropogenic as well as wildfi emissions

(RETRO)			world.,	
Regional Air Pollution Information and Simulation (RAINS)	IIASA, 2004		It combines information on economic and energy developm emission control potentials and costs, atmospheric dispersic characteristics and environmental sensitivities towards air pollution in order to estimates the impacts on human health ecosystems. These expected impacts can then be compared with environmental targets, highlighting areas where the assumed measures fail to meet the environmental policy objectives.	six Kyoto gases (CO_2 , CH_4 , N HFC, PFC and SF6)
Greenhouse Gas and A Pollution Interactions a Synergies (GAINS)	IIASA, 2008		It provides a consistent framework for the analysis of co-benefits reduction strategies from air pollution and greenho gas sources.	CO_2, CH_4, NO_x, N_2O, PM, SO_2 VOC . Certain versions of the GAINS Model also contain: NH_3, CO, Fluorinated greenhouse gases (Gases)

One of the first attempt to estimate the GHG emissions of large cities was the project called “The urban CO_2 Project” (Harvey, 1993), through which 14 cities were involved to develop CO_2 emission inventories, and strategies for its reduction. Other cities as Toronto, Canada (City of Toronto, 1991) and Bologne, Italy (Comune di Bologna, 1995) followed this initiative. The City of Barcelona, Spain, also estimated GHG emissions from 1987 to 1996 (Baldasano et al., 1999). Gases that had most incidence on the greenhouse effect were CO_2 and CH_4. The main sources of these non-biogenic emissions were analyzed and results showed that vehicle traffic represented the principal source of GHG (30-35% of total emissions). The use of natural gas affected total CO_2 emissions for 12% and liquefied petroleum gases (as propane and butane) for 4%. The production of electricity was responsible for approximately 13 %, whereas the Municipal Solid Waste (MSW) disposals (as incinerators and landfills) represented the second principal source of total CO_2 emissions (20-25%). The total emission of CO_2 per inhabitants per year was about 3.4t, low compared to those of other cities (Table 3.8). Emissions of CO_2 from Sydney were about 7 t cap^{-1} yr^{-1} in 1970s and increased to 9 t cap^{-1} yr^{-1} in 1990s (Newman, 1999). In Hong Kong, in 20 years, the emission of CO_2 was almost doubled from 2.3 t cap^{-1} yr^{-1} (Newcombe et al., 1978) to 4.8 t cap^{-1} yr^{-1}(Warren-Rhodes and Koenign, 2001). In London, in the late 1990s, CO_2 emissions varied from 5.5 to 8.5 t cap^{-1} yr^{-1} (Kennedy et al., 2007).

Table 3.8. Comparison of CO_2 emissions per capita in different cities (from Baldasano et al., 1999).

	t CO_2 cap^{-1} yr^{-1}
Barcelona (E), 1996	
Ankara (TK), 1988[a]	
Bologne (I), 1988[a]	
Copenhagen (DK), 1988[a]	
Heidelberg (D), 1987[b]	
Helsinki, (Fin), 1988[a]	
Turin (I), 1990[c]	
Sanjosè, CA (USA), 1988[a]	
Portland, OR (USA), 1988[a]	1
Saarbrucken (D), 1988[a]	1
Hanover (D), 1988[a]	1
Dade county, Miami, FL (USA), 1988[a]	1
Toronto metro, on (Can), 1988[a]	1
Toronto city, on (Can), 1988[a]	1
Minneapolis, MN (USA), 1988[a]	1
Denver, Co (USA), 1988[a]	2

a Harvey, 1993
b Schmidt, 1994
c SOFTECH Energia Tecnologia Ambiente, 1989

About pollutant, in the majorities of cities in developed countries, local high level of sulphur dioxide and particulate matter have decreased for two reasons: the use of natural gas for domestic heating instead of coal and oil, and the relocation of the industries out of the cities. The amount of nitrogen oxides and CO_2 are, instead, increasing or at a critical level.

The Figure 3.5 shows the contaminant emissions from selected cities (Kennedy et al., 2007). Hong Kong, China, and Sydney, Australia, showed a decrease in SO_2 emissions, but an increase in NOx since the 1970s. Also in Korean cities, SO_2 concentration has decreased while CO and NO_2 have not decreases due to continuous use of fossil fuel sources of energy and sharp increase in the number of automobiles (Yoon and Lee, 2003). Emissions of VOCs are quite similar for Toronto in 1997, Sydney in 1990, Brussels, Belgium, in the early 1970s, and the average U.S. city in 1965, even if Sydney showed the highest level and Hog Kong the lowest level. Data of past two decades showed that particulate matter is significantly lower than the 0.05 t cap^{-1} for the average 1965 U.S. city.

At regional and local scale, an example of emission inventory is that developed for the Lombardy Region, highly industrialized area of about 9 million inhabitants in the north of Italy (Caserini et al., 2001). The inventory is based on a database named INEMAR (INventario EMissioni in ARia), and considers about 220 activities and 12 pollutants (SO_2, NOx, NMVOC, CH_4, CO, CO_2, NH_3, N_2O, TSP, PM_{10}, $PM_{2.5}$, PCDD/Fs) for each of the 1546 municipalities of Lombardy. An overview of the emissions for major pollutants and groups of activities is represented in Table 3.9. Most SO_2 emissions derive from combustion plants in the energy industry (68% of total SO_2 emissions), whereas NOx most important source, road transport, accounts for about 52% of total NOx emissions. NMVOC emissions derive mainly from solvents use (48%) and road transport (23%); road transport is also the most significant source of CO emission (60%). CH_4 (49%), N_2O (68%) and NH_3 (97%) emissions are almost entirely due to agriculture and manure management. PM_{10} and $PM_{2.5}$ emissions, as calculated by the in-depth review of available emission factors, derive mainly from road transport (39% of total PM_{10} emission), while energy production, residential and industrial combustion processes account for another 40%.

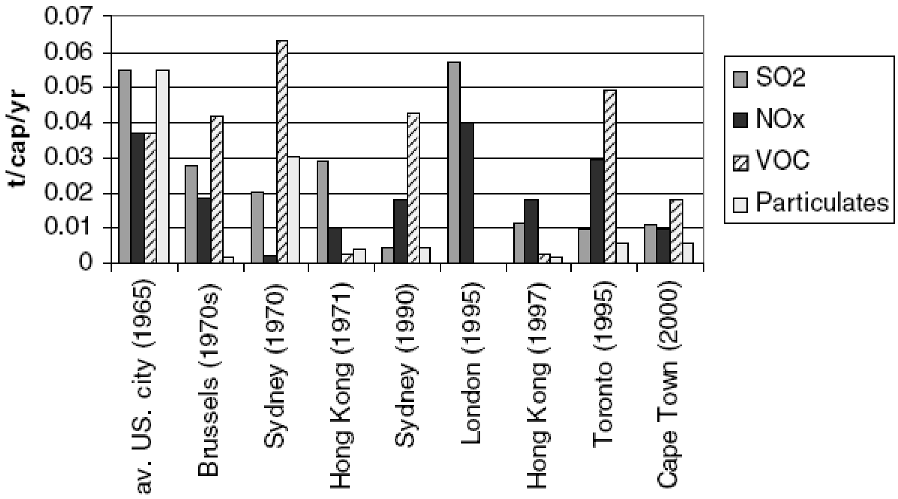

Figure 3.5: Contaminant emissions from selected cities (t cap^{-1} yr^{-1}) (Kennedy et al., 2007).

In scientific applications, where higher resolutions are needed, geographical information such as population densities, land use or other data can provide tools to disaggregate the national level emissions to the required resolution, matching the geographical resolution of the model. Similarly, national emission inventories provide total emissions in a specific year, based on national statistics. In some model applications higher temporal resolutions are needed, for instance when modeling air quality problems related to road transport. In such cases, data on time dependent traffic intensities (rush hours, weekends and working days, summer and winter driving patterns, etc.) can be used to establish the required higher temporal resolution. Emission inventories have to be validated with air quality data. Some factors need to be taken into account: the effective heights of emission influence the relationships between emission rate and ground level concentration; the time of the day and seasonal factors can alter the relative importance of particular sources, and transformations can produce nitrogen dioxide and ozone and deplete chemically reactive and photochemically labile species (Derwent et al., 1995).

Table 3.9: Emissions in Lombardy Region for 2001 (t y^{-1} unless CO_2 in kt y^{-1} and PCDD/Fs in gTEQ y^{-1}) (from Caserini et al., 2001)

	SO_2	NOx	VOC	CH_4	CO	CO_2	N_2O	NH_3	PM10	PM2.5	TSP	PCDD/Fs
1-Combustion in energy and trasformation industries	51.456	26.996	689	710	2.064	16.404	273	1,8	1.450	1.047	1.863	0,3
2-Non-industrial combustion plants	6.244	16.977	13.946	7.720	163.709	16.931	1.837	209	4.733	4.356	5.040	11
3-Combustion in manufactumg industry	10.304	44.554	5.064	1.121	59.114	13.688	743	12	1.846	1.422	2.707	42
4-Production processes	3.524	1.594	30.173	138	24.205	3.890	19	73	2.228	993	2.565	25
5-Extraction and distribution of fossil fuels and geothermal energy			8.919	92.143								
6-Solvent and other product use	1,4	306	148.410		1,6			16	268	87	318	
7-Road Transport	2.913	114.151	72.704	3.079	434.646	18.794	1.740	2.504	8.480	7.739	9.704	4,1
8-Other mobile sources and machinery	1.202	11.320	1.697	32	5.254	969	246	1,6	85	80	88,7	
9-Waste treatment and disposal	271	1.778	268	116.917	150	813	147	3,7	61	55	64	5,2
10-Agriculture		1.568	1.301	216.520	23.314	0	10.656	94.823	1.182	1.010	1.684	
11-Other sources and sinks	83	364	27.787	5.688	10.477	0	12	83	549	511	578	0,4
Total	**75.998**	**219.610**	**310.957**	**444.069**	**722.935**	**71.490**	**15.672**	**97.728**	**20.883**	**17.300**	**24.611**	**87**

3.6 Modeling

By the late 1970s, several air quality and pollution engineering groups were developing global models inclusive of the dozens of photochemical reactions that must be considered in complete treatments (Dodge, 1989; Atkinson and Lloyd, 1984). A variety of numerical simulations have been developed to examine many aspects of the urban environment. For the most part, however, they have been limited to models of single cities in developed countries, not only because they are more tractable, but because funding for such efforts is easier to procure from local jurisdictions (Decker et al., 2000).

Over the last couple of years several model validation and model intercomparison studies have been carried out in Europe in which several models participated in contrast to the usual single model evaluation studies. The large advantage of such a set-up is that the models are also tested against each other, and that a more open discussion originates in which the strong and the weak parts of the different models are analysed (Hass et al., 2003; van Loon et al., 2004).

Most models work in an Eulerian framework, and almost all models use “real” meteorology, as opposed to the more parameterized “monthly” averaged meteorology of the 1990s. Dentener (2006) estimated that there are approximately 25 global models around. Roughly 70% are Chemical Transport Models (CTM); 30% of the present models are either General Circulation Models (GCM), or directly coupled to GCM output.

On local scale, several approaches are used in air quality modeling, such as empirical, Gaussian and numerical. Differences result in terms of input data requirements, complexity of the treatment of atmospheric processes and computational efficiency. Simple empirical models are able to quickly estimate air pollutant concentration and they are specifically used for assessing air quality on street scale. CAR II (Eerens et al., 1993; Teeuwisse, 2003), GRAM (Fisher and Sokhi, 2000), the Design Manual for Roads and Bridges-DMRB (Boulter et al., 2003), and OSPM (Berkowicz et al., 1996) are some examples.

In the following paragraphs, an overview of the main categories of models used in urban environment is reported. Specific models to simulate carbon fluxes and urban ecosystem productivity will be analyzed first, and then models for pollutant simulation will be described.

3.6.1 Flux models

Models to estimate the carbon balance in urban environment derive from **ecosystem process models** used in natural ecosystems. Vertical carbon fluxes of vegetation and soils are estimated using these models with a detailed representation of ecosystem processes. They can simulate responses of vegetation in the cities and in the urban footprint to urban pollution, enhanced levels of atmospheric CO_2, and to changes in urban climate. They can be applied over larger areas and provide regional to global estimates. Some of them are the BIOME-BGC used to simulate carbon balance of turf grasses of the US (Milesi et al., 2005) and the CASA model by which Imhoff et al. (2004) estimated the effect of urban land cover change on NPP of the USA. These models however completely omit horizontal or vertical carbon fluxes associated with human activities.

Light use efficiency model (Monsi and Saeki, 1953; Monteith, 1977) is used to estimate GPP and NPP of urban vegetation, in addition with remotely sensed data. This method detects recent changes in vertical carbon fluxes in response to climate change and land-cover conversion, but it can not be used to detect carbon emission. A recent application of this model was in Michigan, (USA) to analyze changes in land-cover and GPP from a rural-urban gradient (Zhao et al., 2007).

The study of the effects of land use changes on carbon balance is also possible with the statistical models driven by changes in urban population used by Svirejeva-Hopkins et al., 2004. The model considers the horizontal carbon flux associated with transport of organic matter outside of urban areas, but it doesn't account for effect of climate or vegetation management.

All models described above only focus on one effect (climate change, vegetation, or land-cover conversion). There is the need to integrate most of the prominent aspects of urbanization to have a good estimation of the effects of urban characteristics on carbon cycle.

One of the models capable to identify how multiple environmental factors, in particularly climate variability, population density, and species distribution, impact future carbon cycle prediction across a wide geographical range is the ACASA (Advanced Canopy-Atmosphere-Soil Algorithm) (Pyles *et al.*, 2003) model. It estimates energy and mass fluxes between surface and the atmosphere by treating the surface and associated fluxes as an interconnected system. The atmosphere, the urban surface, and the soil are represented as a multilayer system. The model can be used at regional scale when nested in the meso-scale model WRF. It currently has being tested over urban areas.

Carbon fluxes **related to human activities** can be simulated using models developed in industrial ecology, following the urban metabolism concept (Wolman, 1965): the metabolism of a hypothetical city is quantified from overall fluxes of energy, water, materials, and waste into and out an urban region. In urban metabolism models, the physical and biological processes of converting resources into useful products and wastes are like the ecosystem's metabolic processes. The urban metabolism concept has been applied on energy, material, food, and water, but also the carbon balance can be addressed as an urban metabolic process. Anthropogenic emissions of carbon dioxide are related to energy inputs (Kennedy *et al.*, 2007). Carbon inputs into cities are also related to demand for timber, food, or nutrients, while the outputs are related to composition and treatment of waste products. Therefore, to have a good assessment of urban system's impacts on global carbon cycle, the models have to include both biophysical and human dimensions and their interactions (Figure 3.1). Odum and Barrett (2005) put the theoretical bases for understanding carbon cycle in the urban system, but this approach currently does not yet exist. Up to date, carbon cycle models only simulate carbon fluxes through vegetation-soil component

of an urban system ((Bandaranayake et al., 2003; Qian et al., 2003; Imhoff et al., 2004; Svirejeva-Hopkins et al., 2004; Milesi et al., 2005). Progresses should be made on the integration of human activities and biophysical and socio-economic factors affecting carbon cycle over a city.

3.6.2 Transport and dispersion models

Pollutant emissions and dispersion are simulated by models using different spatial resolution and calculation approaches. Regional scale **Chemical Transport Models (CTM)** are widely applied in Europe for various purposes. As diagnostic tools, they are used to improve knowledge of the chemical and physical behaviour of the atmosphere, to help decision makers on the development of policies and air quality management systems for protection of ecosystems and human health. Models are also important instruments for prognostic, applied to a future situation, in order to evaluate the efficiency of a policy or a taken measure to reduce a certain type of emissions, or for air quality forecast, and consequently human exposure and health effects prevention.

CTM models such as CHIMERE, CMAQ, CAMx, among others, are driven by the meteorological fluxes and variables given by meteorological models as MM5 and WRF. The CTM models referred account not only for gas phase processes and photochemical production, but also for aerosol formation, transport and deposition of both inorganic (sulphate, nitrate, ammonium) and organic species, from primary and secondary origin. They simulate the atmospheric chemistry based on lumped carbon mechanisms (such as CB-IV or CB05 or RADM) and a detailed description of the photochemistry. Input data such as emission data (amount of pollutant emitted in a grid cell per second) are required by CHIMERE and CMAQ to estimate the ozone concentrations (and other secondary pollutants) in the atmosphere. They use different aerosol models to estimate primary and secondary PM concentrations in the atmosphere. CTM models have been applied for different purposes, from the assessment of the air quality in several European cities (Vautard et al., 2007; Thunis et al., 2007), operational air quality forecast (Vautard et al., 2001; Monteiro et al., 2005), to the intercomparison exercises, sensitivity analysis and evaluation of a model itself (Morris et al., 2003; Hodzic et al., 2004, 2006). CMAQ has been and continues to be extensively used by EPA and the states for air quality management analyses (SIPs; CAIR, CAMR, RFS-2 rulemakings), by the research community for studying relevant atmospheric processes, and by the international community in a diverse set of model applications (Appel et al., 2007; 2008).

Historically, air pollution forecasting and numerical weather predictions (NWP) have developed separately (Baklanov et al., 2008). This was unavoidable in previous decades when the resolution of NWP models was too poor for meso-scale air pollution forecasting. Due to modern NWP models which approach meso- and city-scale resolution and the employment of land use databases with finer resolution, this situation is changing. Most CTMs have embedded meteorological pre-processors/drivers or are coupled to one, and currently two types of photochemical systems are distinguished: online and offline.

An example of an online meso-scale modeling system is WRF/CHEM, which simulates simultaneously the meteorology and the chemistry (being closer to the atmospheric reality). A comparison between WRF/CHEM and CMAQ model was performed by Lin and Holloway (2009). Zhang et al. (2009a,b) tested the WRF/ CHEM model from regional to global scale. The system WRF/CHEM uses the same principles than MM5 and WRF, but it includes the chemical solver in every time-step together with the meteorology. Based on an adaptation of MM5-CMAQ is the MICROSYS system (Micro-scale air quality modeling system) (EPA, US, 2002). MICROSYS model is capable to simulate with very fine spatial and temporal resolution the atmospheric flow for wind, temperature and humidity and also to account for the dispersion and reaction of pollutants. It was already applied in Madrid center (San Josè et al., 2004; 2008).

Computational fluid dynamics (CFD) has recently been applied to the modelling of urban dispersion. There are essentially two different approaches to the numerical modelling of dispersion through an urban

area using CFD; namely, the Reynolds-averaged Navier-Stokes (RANS) and large-eddy simulation (LES) approaches. The application of CFD to turbulent dispersion in the urban environment using either RANS or LES include Liu and Barth (2002), Baik et al. (2003), Kim and Baik (2004), Camelli et al. (2005), Coirier et al. (2005), Hsieh et al. (2007) and Milliez and Carissimo (2007). A **hybrid approach**, which uses a RANS-predicted gridded field of (building-resolving) wind statistics in an urban area as input to a three-dimensional Lagrangian stochastic trajectory model for the prediction of urban dispersion, is described by Wilson (2007).

The local scale dispersion model VADIS is a Computational Fluid Dynamic (CFD) model that uses a Lagrangian approach to calculate instantaneous concentrations of non-reactive atmospheric pollutants. The model was improved in order to better describe urban real- world conditions (multi-obstacles, multi-source emission) (Borrego et al., 2000) and is particularly valuable under low-wind speed and varying wind conditions (Borrego et al., 2003). It was applied with the transport emission model for line sources named TREM in Lisbon (Borrego et al., 2003). TREM is designed to support quantification of emissions induced by road traffic and the emission rate is estimated as a function of average speed. Another CFD model used for analysing complex street situations include the micro-scale model MIMO (Ehrhard et al., 2000).

MICROSYS and VADIS models can work in a micro-scale domain and they can include buildings, roads, sidewalks, trees, etc. to simulate the closest urban domain with a 4D interaction between biosphere and atmosphere. They also can be nested with the MM5 and WRF meso-scale models and receive boundary conditions and initial conditions by them. These models require also information related to emission data which is usually produced by a traffic model in an urban context. The traffic model called CAMO, developed by UPM, Spain, is available to predict traffic emissions.

Several fast-response **dispersion models** of varying levels of fidelity have been developed to explicitly account for the effects of buildings. Several are intended for use around a single building so are not directly applicable to neighborhood-scale dispersion problems. Recently, several codes have been developed to treat these scales. An Urban Dispersion Model is a **Gaussian puff model** that utilizes simple algorithms for puff-building interaction (Hall et al., 2000; Hanna et al., 2003). Although the model does not produce wind fields around buildings, it accounts for mixing in the lee of the building and some channeling effects. Hall et al. (1997) describe an *empirical* Gaussian puff model that considers the local interaction of puffs with obstacles, while Williams et al. (2004) described a *semi-empirical* urban diagnostic wind model (QUIC-URB), which is used to provide the necessary velocity statistics for a Lagrangian stochastic model of urban dispersion (QUIC-PLUME).

A **stochastic model** based on neural network was developed and used to predict PM10 concentration in Phoenix (Mammarella et al., 2009). They predict PM10 both using a stochastic model (EnviNNet) coupled with a deterministic model (CMAQ). The neural network is found to better predict moderate to high PM10 concentrations than CMAQ.

Using a **mechanistic air quality model** it is possible to predict the contribution of individual emissions source types to the size- and chemical-composition distribution of airborne particles. The model uses the Lagrangian processes trajectory model that examines the evolution of the size- and chemically resolved ambient aerosol when gas-to particle conversion processes are active. It separated the primary particles emitted from different sources: catalyst and not catalyst-equipped gasoline engines, diesel engines, meat cooking, paved and not paved road dust, and sulphur-bearing particles from fuel burning and industrial processes. This separation helps to visualize the effect that different emissions control programs would have in advance of their adoption. The total particulate emission rate for a specific source along with the size and composition profile assigned to that source is used to create a chemical profile for particles emitted to the atmosphere. This model was used by Kleeman and Cass (1998) in Claremont (California) during the 28 August 1987.

Yee and Wang (2009) proposed the formulation of a **probabilistic model** for prediction of the statistical characteristics of concentration fluctuations in pollutant plumes dispersing in an urban area. The formulation arises from the recognition that concentration fluctuations are critical for the proper assessment of flammable and toxic gases and from increased concerns about releases of these noxious substances in (built-up) urban areas where the population is greatest.

Zheng et al. (2005) used the **chemical mass balance receptor model** (CMB) and Song et al. (2006a) used the positive matrix factorization method to apportion the PM2.5 sources. However, the chemical receptor models assume a linear or bilinear relationship between the source profiles and the ambient concentrations. The dispersion model, on the other hand, using a dynamic method and the emission inventory, could solve the problem and clearly identify pollutant sources, which is knowledge required by policy makers. The CALPUFF modelling system (Scire et al., 2000a, b) was used with meteorological observations and the emission inventory to investigate PM10 dispersion in Beijing in the winter of 2000.

To address the specific problems of regional pollution and waste management, an approach was developed in the 1970s modeling **urban biogeochemical fluxes** (Bower, 1977) relative to Residuals-Environmental Quality Management (REQM). REQM quantifies and models the generation and flow of waste outputs from anthropic processes (i.e. residuals). Residuals' impacts on ambient environmental quality are then determined (Bower, 1977). REQM analysis was integrated with the development of models to predict residuals' production, and to estimate costs of reducing residuals to desired levels. Although no REQM studies have occurred recently, the conceptual approach is carried on in environmental resource management models. It is attractive because it focuses specifically on measuring and managing pollutants and water quality (White, 1994).

A model was developed to assess urban sustainability (Yoon and Lee, 2003). **Urban Sustainability Assessment Model** (USAM) is composed of three basic models: driving force, pressure, state, effect, response (DPSER) model, stage model, and the space size model. The model was applied on 57 cities of Korea subdivided into three groups depending on the population. This model simulated a scenario on possible changes in the future (2007-2017) and its outcome can be used as basic data to present effective measures, policies, and plans to address the changes.

3.7 Remote Sensing

Satellite remote sensing technique has been increasingly used in the last decade to quantify aerosols over the globe (King et al., 1999). One of the main sectors that get an advantage from this technique is the modelling of urban environment. In fact, information acquired by remote sensing technique constitutes the input basal layers, in two or three dimensions, used by models to simulate urban phenomena.

Remote sensing refers to the use of electromagnetic radiation to acquire information without being in physical contact with the object, which in this case is the atmosphere. Remote sensing instruments do not directly measure atmospheric composition. Rather, retrieval is conducted by calculating the atmospheric composition that best reproduces the observed radiation. Such retrievals often require external information on geophysical fields. The development of a variety of algorithms to extract physical parameters by accounting for atmospheric radiative transfer has been integral to the success of modern remote sensing.

Satellite sensors provide global images of the Earth and allow retrieving the spatio-temporal aerosol distribution, which results from the spatial inhomogeneities and the short aerosol lifetime (Remer et al., 2005; Santese et al., 2007). Thus the aerosol remote sensing from long-term operational satellites provides a unique opportunity to achieve a global and seasonal monitoring of aerosol load and properties. Kaufman et al. (2002) underlined the important role of satellite sensors in providing the much needed aerosol information for global climate studies. Some disadvantages in the use of satellite sensor

techniques consist of the low temporal resolution, the effects of surface albedo and the presence of clouds.

In order to reduce the uncertainties in aerosol measurements, various networks have been established (e.g., Aerosol Robotic Network (AERONET), Global Atmosphere Watch (GAW), European Aerosol Lidar Network (EARLINET)), which aim to improve the scientific knowledge on aerosol properties and their impact on climate.

Ground-based measurements such as those provided by the AERONET network (Holben et al., 2001) generate key optical parameters such as aerosol optical depth (AOD) at high temporal frequency (every 15 min) but at coarse spatial resolution, while satellite data provide better spatial coverage, but at coarse temporal resolution (typically one image per cloudless day for aerosol remote sensing sensors).

Satellites measure the angular dependence of polarisation and radiance in multiple wavelengths in the UV through the IR at fine temporal and spatial resolution. From these observations, retrieved aerosol products include not only optical depth at one wavelength, but spectral optical depth and particle size over ocean and land, as well as more direct measurements of polarisation and phase function. Examples of such new and enhanced sensors include Polarization and Directionality of the Earth's Reflectance (POLDER), MODIS, and Multi-angle Imaging Spectro-Radiometer (MISR), among others. Aerosol profiling from space is also making promising progress (Yu et al., 2006).

Extensive analysis and comparison of satellite retrievals over central Europe has recently been published by Kokhanovsky et al. (2007). Table 3.11 describes the satellite retrievals differences in terms of spatial and temporal resolution, platform used and angle of observation. Although these satellite instruments use different algorithms should ideally produce consistent values for the aerosol properties for a given scene.

Monitoring aerosols from space has been performed for over two decades (King et al., 1999). There is a relatively long history of the quantitative estimation of aerosol optical depth (AOD) from remotely sensed imagery using multiangular information (Diner et al., 2005; North, 2002), polarization information (Deuze et al., 2001), multispectral information (Kaufman et al., 1997; Liang and Fang, 2004; Liang et al., 1997; Teillet and Fedosejevs, 1995) and multitemporal information (Christopher et al., 2002; Hauser et al., 2005; Zhang and Christopher, 2001). Several studies have attempted to use the aerosol optical thickness (AOT) retrieved from satellite imagery to monitor aerosol loading and the associated air quality effects (Kaufman and Fraser, 1983; Fraser et al., 1984).

The Terra satellite (December 1999) significantly expanded scientific perspective about the scale of tropospheric pollution. The MOPITT instrument onboard Terra is a nadir-viewing gas correlation radiometer operating in the 4.7 μm band of carbon monoxide (Drummond and Mand, 1996). The MOPITT pixel is 22x22 km^2 at nadir with a 29 pixel wide swath.

The data sets from the recently launched MODIS (on Terra and Aqua satellites) provide an unprecedented opportunity to monitor aerosol events and examine the role of aerosols in the Earth-atmosphere system (Kaufman et al., 2002). The MODIS sensor has the ability to characterize the spatial and temporal characteristics of the global aerosol field. MODIS has 36 channels spanning the spectral range from 0.41 to 15 μm representing three spatial resolutions: 250 m (2 channels), 500 m (5 channels), and 1 km (29 channels). The aerosol retrieval makes use of seven of these channels (0.47–2.13 μm) to retrieve aerosol characteristics and uses additional wavelengths in other parts of the

Table 3.10: CO_2 and air quality models, by scale from regional to micro-scale

Model	**SCALE**	**Processes**	**References**
CHIMERE	Regional scale	Parabolic Piecewise method (PPM), Godunov scheme and the simple upwind first-order scheme	Bessagnet et al., 2004; Hodzic et al., 2004; 2006; Monteiro et al., 2007
CMAQ	Regional scale	It uses different aerosol models to estimate primary and secondary PM concentrations in the atmosphere.	EPA (US); Mammarella et al., 2009
CAMx	Regional scale	Eulerian continuity equation closed by K-theory equations expressed in flux form	Vautard et al., 2001; 2007
TAPM	Regional scale	Semi Langrangian approach	Hurley, 2000; Hurley et al., 2005a,b
CALGRID	Regional scale	Finite difference scheme based on cubic splines. Stability dependent, Smagorinsky, or both schemes for turbulent diffusion coefficients	Yamartino, 1993; Yamartino et al., 1989; 1992; 1996
WRF/CHEM	Regional scale	On-line model which simultaneously simulates the meteorology and the chemistry (being closer to the atmospheric reality)	Zhang et al., 2009a, b
EnviNNet	Regional scale	Stochastic model based on neural network	Mammarella et al., 2009
USAM	Regional scale	Three basic models: driving force, pressure, state, effect, response (DPSER) model, stage model and space size model	Yoon and Lee, 2003
CAMO	Local scale	Traffic model (for urban environments) based on a cellular automata model	San Josè et al., 2006
QUIC-PLUME	Local scale	Uses a stochastic Lagrangian random -walk approach to estimate concentrations in a 3D gridded domain.	Williams et al., 2004
CHEMICAL MASS BALANCE (CMB) MODEL	Local scale	Consists of a solution to linear equations that express each receptor chemical concentration as a linear sum of products of source profile abundances and source contributions.	Zheng et al., 2005; Srivastava and Jain, 2007
CALPUFF	Local scale	Dispersion model	Scire et al., 2000a, b; Song et al., 2006b
REQM	Local scale	It models the generation and flow of waste outputs from anthropic processes	Bower, 1977
BIOME-BGC model	Local scale	The model requires daily climate data and the definition of several key climate, vegetation, and site conditions to estimate fluxes of carbon and nitrogen.	Milesi et al., 2005
HPAC (Hazard Prediction and Assessment Capability)/ SCIPUFF	Local scale	A Lagrangian puff dispersion model that uses a collection of Gaussian puffs to represent an arbitrary, three-dimensional, time-dependent concentration field	Hanna et al., 2009
CAR II	Local and street scale	Simple empirical model	Eerens et al., 1993; Teeuwisse, 2003
GRAM	Local and street scale	Only analytical formulae	Fisher and Sokhi, 1998
DMRB	Local and street scale	Simple empirical model	Boulter et al., 2003
OSPM	Local and street scale	Simple empirical model	Berkowicz et al., 1996
VADIS	Local and street scale	CFD model using a Lagrangian approach to calculate instantaneous concentrations of non-reactive atmospheric pollutants.	Borrego et al., 2003; 2006
MICROSYS	Local and street scale	Micro-scale fluid dynamics model which includes chemical dispersion and transformation of species.	San Josè et al., 2004; 2008
TREM	Local and street level	Transport emission model for line sources	Borrego et al., 2003
MIMO	Local and street level		Ehrhard et al., 2000
ACASA	Local and regional scale when coupled with WRF	Higher-order closure model	Pyles et al., 2003

Table 3.11: Characteristics of selected satellite instruments used to produce aerosol optical properties (from Kokhanovsky et al., 2007).

Instrument	Satellite/time of measurement	Swath (km)	Channels	Spatial resolution	Multi-angle observation
MERIS	ENVISAT 10:00 UTC	1150	15 bands 0.4–1.05 μm (0.41,0.44,0.49,0.51,0.56, 0.62,0.665,0.681,0.705,0.754, 0.76,0.775,0.865,0.89,0.9 μm)	0.3×0.3 km^2	No
AATSR	ENVISAT 10:00 UTC	512	7 bands 0.55,0.66, 0.87, 1.6, 3.7, 10.85, 12.0 μm	1×1 km^2	Yes, 2 angles from the ranges 0–21.732 and 55.587–53.009 degrees
SCIAMACHY	ENVISAT 10:00 UTC	916	8000 spectral points 0.24–2.4 μm	30×60 km^2	No
MISR	TERRA 10:32 UTC	400	4 bands 0.446, 0.558, 0.672, 0.866 μm	0.25×0.25 km^2 at nadir and at 0.672 μm 1.1×1.1 km^2 in the remaining channels	Yes, 9 angles 0, 26.1, 45.6, 60.0, 70.5°
MODIS	TERRA 10:32 UTC AQUA 13:30 UTC	2300	36 bands 0.4–14.4 μm (1):0.659,0.865 (2):0.47,0.555,1.24,1.64,2.13 (3):0.412,0.443,0.488,0.531,0.551, 0.667,0.678,0.748,0.869,0.905,0.936, 0.94,1.375+MWIR(6)/LWIR (10) channels	(1): 0.25×0.25 km^2 (2): 0.5×0.5 km^2 (3): 1×1 km^2	No
POLDER	PARASOL 13:33 UTC	1700	8 bands 0.443,0.490*,0.565,0.670*, 0.865*,0.763,0.765,0.91	5.3×6.2 km^2	Yes, channels marked with* have a capability to measure polarization

spectrum to identify clouds and river sediments (Ackerman et al., 1998; Gao et al., 2002; Martins et al., 2002; Li et al., 2003). Unlike previous satellite sensors, which did not have sufficient spectral diversity, MODIS has the unique ability to retrieve aerosol optical thickness with greater accuracy and to retrieve parameters characterizing aerosol size (Tanré et al., 1996; Tanré et al. 1997).

The MODIS aerosol algorithm is actually two entirely independent algorithms, one for deriving aerosols over land and the second for aerosols over ocean. Both algorithms are described in depth in Kaufman et al. (1997) and Tanré et al. (1997). In addition, Levy et al. (2003) provide a more recent description of the over ocean retrieval algorithm. Both the land and ocean aerosol algorithms rely on calibrated, geolocated reflectances provided by the MODIS Characterization Support Team (MCST), identified as products MOD02 and MOD03 for Terra MODIS products and MYD02 and MYD03 for the Aqua MODIS products (MCST 2000, 2002). The uncertainties in these measured reflectances in the visible and mid-IR bands are less than 2%

There are few alternative algorithms. Hsu et al. (2004) proposes a new approach to retrieve aerosol properties over bright surfaces using the minimum reflectance determined from 0.1°× 0.1° grid. The spatial resolution is not sufficient for atmospheric correction of MODIS imagery at 500 m or 1 km scale. The accuracy of 30% at this scale also needs to be improved. Tang et al. (2005) establishes an empirical relationship between the top of atmosphere reflectance and surface reflectance, and MODIS data from two successive orbits are used to solve the nonlinear equation. Any errors resulting from geometric registration and subpixel clouds may significantly affect the solutions. Liang et al. (2006) developed a new algorithm for estimating the aerosol optical depths using multi-temporal MODIS data over land surfaces. The algorithm was validated using AERONET measurements. Therefore, further development of new algorithms is critically needed, especially when aerosol information is required in hot spot areas with a greater spatial resolution.

Most of urban studies use data and optical parameters such as the aerosol optical depth (AOD) and thickness (AOT) retrieved by MODIS, POLDER, MISR, and Lidar observations. Several studies analyzed different remote sensing approaches and compared them.

Vachon et al. (2004) compared the polarized ADEOS-1 POLDER measurements with the MODIS dark target retrieval over North America landscapes. Aerosol optical thickness (AOT) data retrieved by the Multi-angle Imaging SpectroRadiometer (MISR) from 2002 to 2004 were compared with AOT measurements from an Aerosol Robotic Network (AERONET) site located in Beijing urban area (Jiang et al., 2007), showing good correlation. Besides, observations by MODIS and measurements of pollution in the troposphere (MOPITT) were taken in the vicinity of Mexico City to illustrate current satellite capabilities (Massie et al., 2006). Satellite data from AOD, True Color Images from MODIS and vertical aerosol profiles from CALIPSO were used in addition to PM2.5 measurements for analysing air quality events in the state of Georgia (Alston and Sokolik, 2008). In the city of Florence (Italy), a study to analyze the daily cycle of aerosol was performed using an unattended light detection and ranging (LIDAR) operating at 532–1064nm. (Del Guasta, 2002). This was the first time that this technique was used in Italy for long-term monitoring of urban aerosol. The observed cycles were the result of a coupling between the traffic cycle and the daily surface-wind cycle. Global cities were investigated by Gupta et al. (2006). Particulate matter was estimated in 26 locations in Sydney, Delhi, Hong Kong, New York City and Switzerland using 1 year of aerosol optical thickness (AOT) retrievals from MODIS along with ground measurements of PM2.5 mass concentration. This study is among the first to examine the relationship between satellite and ground measurements over several global locations, and excellent correlation was found.

In the proximity to the metropolitan regions of Dallas and Houston (Texas) (Hutchison, 2003; Hutchison et al., 2004), data from transient pollution moving from the central US to Texas, were re-analyzed in an attempt to predict air quality from MODIS aerosol optical thickness (AOT) observations. The results demonstrate a method to forecast air quality from remotely sensed satellite observations when the transient pollution can be isolated from local sources.

Ozone concentration was also investigated by Kim et al. (2007) in Seoul, Korea. Three ground-based active remote sensing instruments operated side by side: micro-pulse lidar (MPL), differential absorption lidar (DIAL), and differential optical absorption spectroscopy (DOAS). A long-term study was conducted over Athens (Greece) from 2000 to 2005 with the aim to assess the inter-annual and seasonal variability of the aerosol properties over the city (Kosmopoulos et al., 2008).The data set of aerosol optical properties was obtained from MODIS. Three aerosol types were identified (urban/industrial aerosols, coarse-mode particles and clean maritime conditions) and the coarse-mode particles exhibited much stronger interannual and seasonal variability compared to the urban/industrial aerosols.

3.8 Future-Conclusion

Additional studies of the carbon balance of settlements of varying densities, geographical location, and patterns of development are needed to quantify the potential impacts of various policy and planning alternatives on net greenhouse gas emissions. Different aspects of urban form (e.g., housing density, availability of public transportation, type and location of forest cover) may have different net effects on carbon sources and sinks, depending on the location, affluence, economy, and geography of various settlements. It is possible to develop quantitative tools to take many of these factors into account. To facilitate development and application of integrated urban carbon cycle techniques and to extrapolate local studies to regional, national, and continental scales, useful additional data include:

- common land cover classifications appropriate for characterizing a variety of human settlements
- emissions inventories at small spatial scales such as individual neighbourhoods and municipalities

- expansion of the national carbon inventory and flux measurement networks to include land cover types within human settlements
- comparative studies of processes and drivers of development in varying regions and nations
- interdisciplinary studies of land-use change that evaluate socioeconomic as well as biophysical drivers of carbon sources and sinks

In general, there has been a focus in carbon cycle science on measuring carbon concentration and fluxes in natural ecosystems, and consequently highly managed and human-dominated systems such as settlements have been underrepresented in many regional and national inventories. To assess the full carbon balance of settlements ranging from rural developments to large cities, a wide range of measurement techniques and scientific, economic, and social science disciplines are required to understand the dynamics of urban expansion, transportation, economic development, and biological sources and sinks. An advantage to an interdisciplinary focus on the study of human settlements from a carbon cycle perspective is that human activities and biological impacts in and surrounding settled areas encompass many aspects of perturbations to atmospheric CO_2, including a large proportion of national CO_2 emissions and changes in carbon sinks resulting from land-use change.

No unique urban picture emerges based on the currently available of atmospheric fluxes reported in the literature. The site to site flux variability reflects the real diversity of urban areas. This diversity is not yet adequately covered by experimental studies. Putting aside instrumental and methodical issues, the fluxes are valid for the point in space where they were measured. It is well known that fluxes measured at a certain height are influenced by a certain source area. The extent of this area needs to be known, and detailed descriptions of surface properties are necessary (e.g. the vegetation fraction) in order to be able to better assess the influence of biospheric uptake or release of CO_2 and trace gases. Clearly, more long-term studies from a variety of cities are needed.

Some efforts are needed to develop a common operational air quality forecasting model with fully coupled physical and chemical processes. The creation of a common predictive modelling platform for parameterization testing, evaluation, and effective integration of research contributions will be very useful. The evaluation of model results is vital to the improvement of models. A study was conducted by Angevine et al. (2009) to evaluate the results of meso-scale Eulerian numerical model runs to decide whether changes to the modelling system improve results for air quality applications. The Advanced Research core of the Weather Research and Forecasting model (WRF-ARW) model system was run for 75 days at 5km grid spacing. Results from shorter runs with a finer grid have been shown by Angevine et al. (2008). The authors found that assimilation of wind profiler data improves the model results overall, primarily by reducing the random error in wind direction. The improvement was most easily seen in the wind profile away from the surface, and confirmed by a tight correlation with measured ozone.

Beside, air quality models need to be properly evaluated before their predictions can be used with confidence, since model results often influence decisions that have large public health and economic consequences. Therefore, information about uncertainties should be correctly estimated and interpreted since it is as important as modelling data.

The uncertainty concept is one of the crucial points of Quality Assurance/Quality Control (QA/QC) procedures that should provide quantitative information about the modelling precision, identifying the sources of uncertainty and their potential reduction. The current European legislation defines the requirements of QA/QC procedures for air quality modelling, including the definition of quality objectives as an acceptability measure, in order to guarantee a good model performance and reliable modelling results for decision makers (Borrego et al., 2008).