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URBAN SEDIMENTS—A GLOBAL PERSPECTIVE • REVIEW ARTICLE

Sediments in urban river basins: a review of sediment–contaminant dynamics in an environmental system conditioned by human activities

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Abstract
Background, aim and scope Over 50% of the global population live in urban centres and, therefore, an understanding of the processes acting upon urban systems is a global issue. The nature of human-made, often impervious, land surfaces and heavily engineered waterways results in hydrological and sedimentological systems in urbanised basins which contrast significantly to those within more natural (i.e. pristine, forested, agricultural) aquatic systems. In addition, the abundance of contamination sources in urban systems results in chemical pressures often manifested as high pollution concentrations or loadings, which in turn have detrimental impacts on human and ecosystem health. These lead to management and sustainability issues not generally encountered in more natural environments. The purpose of this review is to provide a state-of-the-art assessment of sediment sources, pathways and storage within urban river systems, to consider sediment management within urban systems and river basins, and examine the role of local and global environmental changes on sediment processes and management. Inevitably, much of the sediment that is transported within urbanised basins is contaminated, so this review also considers sediment–contaminant sources and interactions.
Conclusions and recommendations We reach a number of conclusions and recommendations for future research. There is a need for better sampling and monitoring of sediment and sediment-associated contaminant fluxes and cycling in urban river channels and basins. This should include better techniques and studies to identify sources and transfers of road-deposited sediment (RDS), airborne particulate matter and sediments in the river system. Greater interdisciplinary research, combining sedimentologists, hydrologists, urban planners, urban archaeologists, chemists and biologists, is needed. More attention needs to focus on upscaling and connecting urban areas to the rest of the river basin, both upstream and downstream. Finally, there is a need to balance multiple needs (urban population, water resources) with likely trends in both urban development and global environmental change.

Keywords Contaminants · River basin · Sediment · Sediment management · Urban · Urbanisation

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1 Background, aim and scope

At the time of writing, over 50% of the global population live in urban centres, with this value set to increase (UNFPA 2007). Thus, for example, about 80% of the UK population live in conurbations, and even in a large, relatively unpopulated country such as Canada about 70% of the population live in urban areas. In addition, the number of 'mega-cities' (those cities with a population of greater than 10 million) is also increasing, particularly in the developing world: China, India and South America. Therefore, an understanding of the processes acting upon urban systems is a global issue. The nature of human-made, often impervious, land surfaces and heavily engineered waterways results in hydrological and sedimentological processes in urbanised basins which contrast significantly to those within more natural (i.e. pristine, forested, agricultural) aquatic systems. In addition, the abundance of contamination sources in urban systems results in chemical pressures often manifested as high pollution concentrations or loadings (Duh et al. 2008; Horowitz and Stephens 2008; Laidlaw and Filipeli 2008; Wong et al. 2006), which in turn have detrimental impacts on human and ecosystem health. These lead to management and sustainability issues not generally encountered in more natural environments.

Significant effort has been focussed on understanding surface and sub-surface hydrology, and the chemical quality of surface and groundwater, in urban river basins. Chemicals such as metals and persistent organic pollutants, which by their very presence or by their elevated concentrations represent contaminants to aquatic systems that are transported in dissolved or particulate form, have received considerable attention within urban river systems (Behrendt 1993; Foster and Charlesworth 1996; Salomons and Förstner 1984). Most attention has focussed on concentrations and fates of contaminants in dissolved form (see Salomons and Förstner 1984), with much less attention given to sediment-associated contaminants. However, the important role that sediments play in contaminant transfer and water quality in rivers has recently been recognised by a number of studies (for reviews, see Bilotta and Brazier 2008; Owens 2008; Owens et al. 2005; Viers et al. 2009), and the importance of incorporating sediment into water quality management plans has been recognised (Brils 2004; Förstner 2002). Furthermore, the need to consider sediment management at the river basin scale has been persuasively documented (Apitz and White 2003; Owens 2005, 2008), driven, in part, by the introduction of legislation and policy, such as the EU Water Framework Directive (WFD), that mainly relate to water quality and aquatic ecosystem health within river basins (Casper 2008). For example, in the stepwise implementation strategy of the WFD, the Advisory Forum on Priority Substances has proposed the addition of the specific source/pathway of 'historical pollution from sediments' in the Program of Measures (POM). In addition, the European Parliament, Council and Commission have suggested that EU Member States may opt to apply environmental quality standards (EQS) for sediment and/or biota instead of those for water. It is recognised that the treatment of sediment issues will be the responsibility of river basin communities; these are the central institutions for promoting the development of river basin management plans (RBMPs) and for implementing the POM in their respective river basins, including the handling of complex urban systems. It is clear, therefore, that an understanding of the characteristics and behaviour of sediments within urban systems is crucial for successful implementation of these environmental policy measures.

The purpose of this review is to provide a state-of-the-art assessment of sediment sources, pathways and storage within urban river systems, and to consider the interaction between sediments and associated contaminants within such environments. Unlike most studies to date, which largely focus on urban areas in relative isolation to the wider landscape, here attention is focussed on the role of urban systems as part of the river basin, given the importance of this landscape unit for water and sediment management. We also consider sediment management within urban systems and river basins, and the role of local and global environmental changes on sediment processes and management. Finally, some recommendations for future research and management needs are presented.

2 The urban sedimentary system: definitions, sub-systems and the urban sediment cascade

The term *urban* is used widely throughout both the social and scientific literatures. A definition suggested by Taylor (2007) for *urban areas* is "those areas where the ecosystem is significantly modified by [dense] human settlement and associated activities." This definition distinguishes such urban areas from agricultural areas, because although the latter have human activities they are not typically associated with dense settlement (i.e. buildings, roads and infrastructure). As such, urban areas are characterised by a high degree of modification of the physical, chemical and biological environment, resulting from the construction of buildings at a large spatial scale. The term *urban sediment* is used in a range of different contexts throughout the literature. In the past, *urban sediments* has been a term commonly used to refer to sediment accumulations on street surfaces (Taylor 2007); in this review, the term is used in a broader manner to mean any sediments present within the *urban environment*. While urban soil represents an important source of sediments and airborne particles, we do not consider urban soils in detail here as these are reviewed elsewhere (e.g. Bullock and Gregory 1991; Lehmann and Stahr 2007; Norra and Stueben 2003).

Sediments in urbanised river basins can be broadly categorised into two main types—those deposited upon road surfaces and acted upon primarily by sub-aerial processes and those deposited and transported in aquatic environments (i.e. rivers, canals and docks). These sub-systems are linked through the urban sediment cascade (Fig. 1; Charlesworth and Lees 1999; Taylor 2007). This urban sediment cascade recognises the highly dynamic nature of the urban sediment system and the key relationships

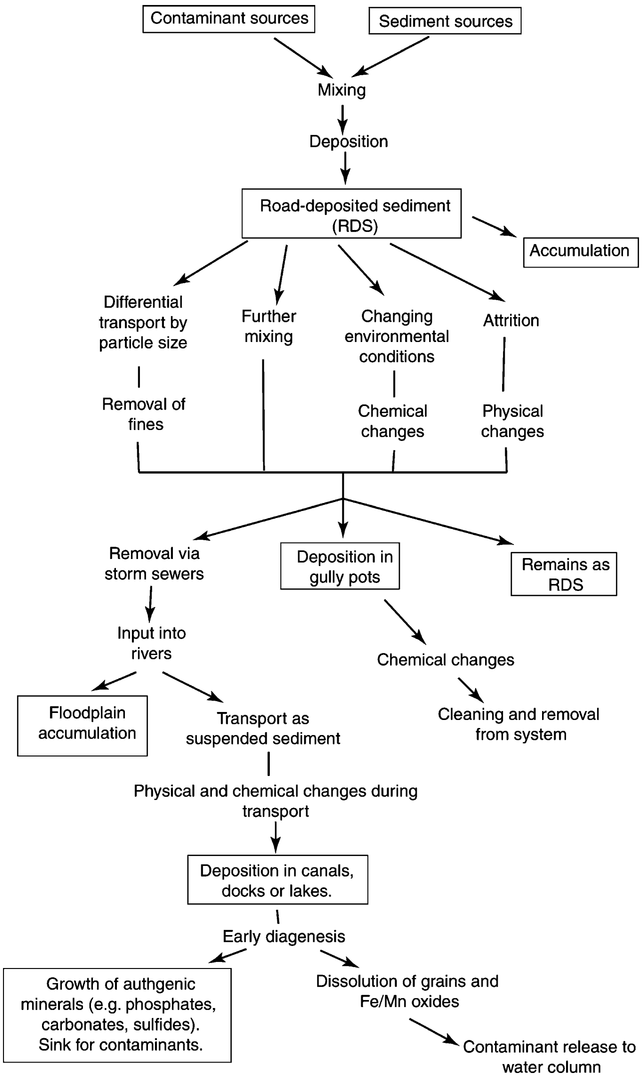

Fig. 1 The urban sediment cascade (from Taylor 2007; reproduced with the permission of Wiley-Blackwell)

between sediment sources, transport mechanisms, deposition (storage) and post-depositional modification of sediment. The major sites of sediment deposition in the urban sediment cascade are: street surfaces; gully pots and storm sewers; rivers; canals and docks; and lakes. Deposition and storage on street surfaces, gully pots and storm sewers is short term, as is the storage of fine-grained (i.e. clays, silts and fine sands) sediment in urban river channel beds; in the order of days to months. As such, these features can be considered as temporary storage elements. Sediment storage in canals, docks and on floodplains, as well as the coarser fraction (coarse sands, gravels etc.) of channel bed sediment, is longer term, and can be in the order of years to decades, although they are not necessarily permanent due to erosion of floodplain banks and dredging of channels, canals and docks. The characteristics of these sediments are discussed further in the sections below.

The transport of sediment through the urban sediment cascade is complex, and there is still a poor understanding of the pathways that sediments take from sources to receiving water bodies, and the rates of sediment transport via these pathways. As in other aquatic systems the bulk of sediment transport in the urban environment takes place through the action of water: the flow of water drives the system and essentially connects the terrestrial part of the urban system to the channel network. However, local redistribution of finer sediment upon street surfaces by wind can take place, and this can have significant impacts upon air quality and human health via airborne particulate matter (see below). Storm drains carry urban runoff from the street surfaces (as well as other impervious surfaces) to receiving water bodies (i.e. rivers, canals or docks). In addition to this surface water runoff, the urban drainage system also has to deal with industrial and domestic wastewaters and sewage. In combined sewer systems, road runoff is transferred to a common sewer, and from then onwards to a sewage treatment works (STW). In this case, these combined sewers rarely have sufficient hydraulic capacity during storm events, and the water is discharged directly (including sewage) through a combined sewage overflow (CSO) outlet into receiving water bodies.

To illustrate the urban sediment cascade and the complexity of the urban sediment system, Fig. 2 provides a schematic illustration of the main sources, transfer pathways and storage features within a hypothetical urban river basin. Many of the interactions illustrated are two-way processes: for example, atmospheric particulates are a key source of road sediment, which, in turn, can represent a source of airborne particulate matter. Unfortunately, there is relatively little information on sediment fluxes and storage volumes, so that it is difficult to construct a typical—if such a thing exists—sediment budget for an urbanised river basin. The sections below review some of the key information available to date.

3 Sources of urban sediment

The sources of sediment and contaminants to river basins are summarised in Table 1, and a full overview can be found in Taylor et al. (2008a). In river basins, the main 'natural' sources of sediments to rivers are: atmospheric dust deposition and wind eroded particles; mass movement events (e.g. landslides and debris flows); and erosion of soils by water and tillage. Additional sources from within the river corridor include: erosion of channel banks; erosion of floodplain deposits; resuspension of channel bed sediment due to changing flow conditions; and erosion of cliffs in coastal areas. All of these may be important in urbanised river basins (see Fig. 2). Sediment sources which

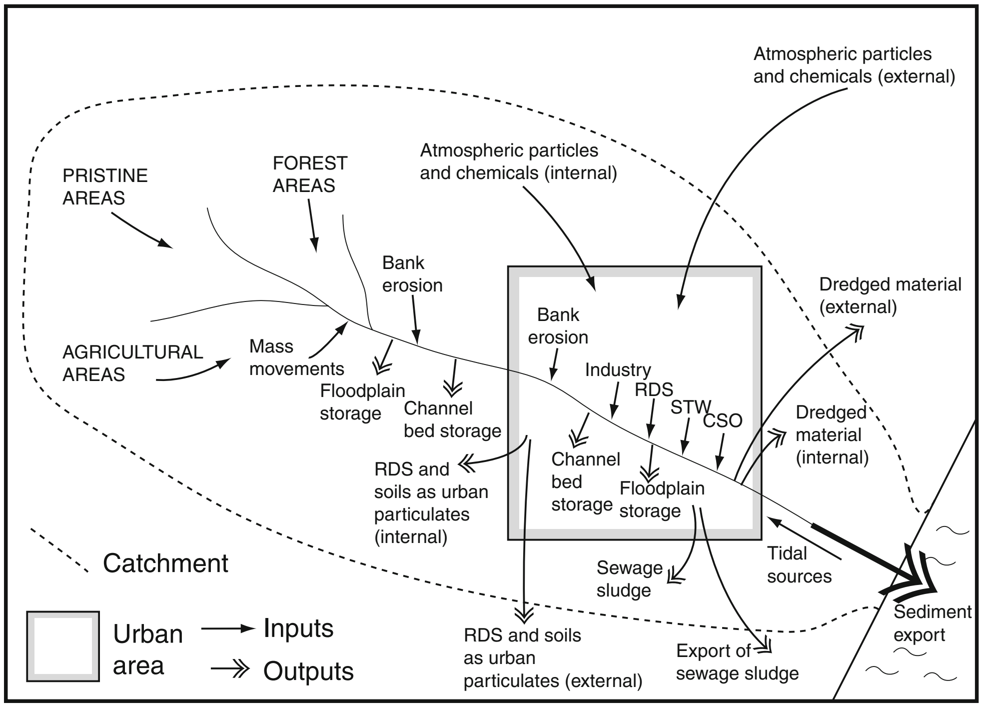

Fig. 2 Conceptual diagram of sediment sources and pathways in an urbanised river basin

are a direct result of anthropogenic activity include: mining; construction; the urban road network; industrial point sources; and wastewater and STWs. Many of these anthropogenic sources dominate (at least the contaminant load) within urbanised river basins.

3.1 Sources to road-deposited sediment

Road-deposited sediment is the accumulation of particulate matter on street surfaces and is abundant in all urban river basins. The term *road dust* has been used extensively in the past, but the term *dust* implies fine material (i.e. grains less that 10 μm) with potential respiratory affects on human health. Numerous studies have shown that street sediment is composed of a full range of particle sizes (Biggins and Harrison 1986; Brinkmann and Tobin 2003; Robertson and Taylor 2007; Sutherland 2003), commonly biased towards coarse material and, therefore, the use of the term *dust* is not appropriate. More recently the term *road-deposited sediment* (RDS) has been used (e.g. Sutherland 2003; Taylor 2007) and is favoured here. RDS has received considerable attention in recent years as a consequence of its potential impact upon urban air quality and urban runoff, its ease of sampling, and its potential to act as a proxy for urban pollution levels.

There is a diverse range of sources to RDS, both anthropogenic and natural. Anthropogenic sources include vehicle exhaust emissions, vehicle tyre and body wear, brake-lining material, building and construction material, road salt, road paint and pedestrian debris, whilst natural sources include soil material, plant and leaf litter, and atmospheric deposition of particles (e.g. Beckwith et al. 1986; Hopke et al. 1980; Lecoanet et al. 2003; Robertson et al. 2003; Thorpe and Harrison 2008; Xie et al. 1999). Importantly, some of the atmospheric sources of particles may derive from outside of the river basin. Volumetrically, the most important component of RDS comes from soils

Table 1 Examples of sources of sediment and associated contaminants to river basins (from Taylor et al. 2008a)

Material	Sources
Sediment (organic and inorganic)	Erosion from rural, agricultural and forested land, channel banks, urban road sediment and construction, STW solids, atmospheric deposition, inputs from tidal areas and coastal zone (during flood and ebb tidal cycle)
Metals and metalloids (Ag, Cd, Cu, Co, Cr, Hg, Ni, Pb, Sb, Sn, Zn, As)	Geology, mining, industry, acid rock drainage, sewage treatment, urban runoff
Nutrients (P, N)	Agricultural and urban runoff, wastewater and sewage treatment
Organic compounds (pesticides, herbicides, hydrocarbons, PCBs, PAHs, dioxins)	Agriculture, industry, sewage, landfill, urban runoff, combustion
Xenobiotica and antibiotics	Sewage treatment works, industry, agriculture
Radionuclides (^{137}Cs, ^{129}I, ^{239}Pu, ^{230}Th, ^{99}Tc)	Nuclear power industry, military, geology, agriculture (secondary source)

and building material (Hopke et al. 1980). Soil material, which may be derived from a range of distances (e.g. farm land on the outskirts of the urban area, and gardens and parks within the urban area), contributes both minerogenic and organic material. Building material contributes quartz sand, concrete and cement to RDS, and these are generally relatively inert with respect to surface water quality.

The majority of contaminants to RDS are derived from anthropogenic sources. Lead is predominantly derived from leaded fuel (where tetra-ethyl lead is used as an additive). Lead levels in sediment have declined, however, with the widespread reduction in use of leaded fuel (Nageotte and Day 1998). Copper and Zn have both been sourced to vehicle activity, with Cu coming from corroded car bodywork (Beckwith et al. 1986), and Zn and Cd being derived from tyre wear (Hopke et al. 1980). Chromium, Br and Mn are also present in tyres and brake linings. Multi-element analysis of RDS, coupled with principal component analysis, has been used in some studies to aid in elemental source apportionment (e.g. De Miguel et al. 1997). The platinum group elements (PGEs), Pt, Pd and Rh, are a relatively recent contribution to RDS, having been emitted into urban environments since the early 1990s as a result of their use as catalysts in catalytic converters. Concentrations of PGEs significantly above those of average upper crust values have been reported for RDS for cities in Europe and Australia (Motelica-Heino et al. 2001; Whiteley and Murray 2003; Wei and Morrison 1994a). Although PGEs in metallic form are generally considered to be biologically inert, soluble PGE salts are indicated to be much more bio-reactive (Farago et al. 1998) and so the presence of PGEs in RDS is potentially of significant concern.

There is a wide range of organic contaminants (so called persistent organic pollutants, POPs) present in RDS. These include polyaromatic hydrocarbons (PAHs), polychlorinated biphenols (PCBs), hydrocarbons, dioxins, pesticides and herbicides. The sources of these are various and include both atmospheric- and land-based sources. For example, PAHs were observed to be sourced from biomass burning and vehicular emissions in Vancouver, British Columbia, Canada (Yunker et al. 2002). Probably, the largest source is that derived from vehicular activity. Many of these are found in petrol or diesel (including benzene, toluene, naphthalene, PAHs), or associated with automobiles (including ethylene glycol, hydraulic fluids, styrene, oil lubricants). Pesticides and herbicides are applied directly to pavements or to urban soils in residential areas and gardens, where they can be removed from runoff or erosion and deposited in RDS.

While atmospheric particles represent an important source of RDS, in turn, RDS (and urban soils) represents an important source of atmospheric fine particulate matter (Mazzei et al. 2008). In many urban 'airsheds', air quality is controlled by both chemical composition and airborne particulate concentration, the latter usually being measured as PM_{10}, $PM_{2.5}$ or PM_1 (particulate material with airborne diameter smaller than 10, 2.5 and 1 µm, respectively). $PM_{2.5}$ and PM_1 are particularly important for human health as such fine particles are able to enter the respiratory system. Studies (e.g. Breed et al. 2002; Rubin et al. 2008) in the city of Prince George, British Columbia, Canada, for example, have used emissions inventory, chemical characterization and source apportionment techniques to demonstrate that road dust is an important source of $PM_{2.5}$ and particularly of PM_{10} (Table 2), with contributions of 10,580 t year^{-1} from unpaved and paved road sediments, and 10,550 t year^{-1} from winter road sanding in 1995 (data in Breed et al. 2002). Other studies (see examples cited in Thorpe and Harrison 2008) have estimated that >50% of the airborne particulate matter may be due to non-exhaust sources such as resuspension of RDS. This illustrates the complex two-way pathways and interactions between atmospheric- and land-based sources of particulate matter and chemicals in urban areas (see Fig. 2).

3.2 Sources of sediment and contaminants to urban rivers

The range of sediment sources for rivers is greater than that for RDS, because most urban river basins comprise a mix of urban and non-urban (i.e. forested, agricultural) land uses. Thus, in addition to the input of RDS into rivers, described above, there are also inputs of sediment derived from the erosion of soil and channel bank material as well as from mass movements (such as landslides) in some basins (see Fig. 2). For example, Göransson et al. (2009) describe the present and historical importance of mass movement events in the urbanised Göta Älv river basin in Sweden. Given the propensity for most urban areas to be in the middle and lower parts of the river basin, the supply of sediment from non-urban land uses is from sources upstream of the urban area because of the downstream flow of water and sediment. In urban areas near the coastal zone, sediment can also be supplied from marine and estuarine sources due to tidal effects (Townsend and Whitehead 2003).

Table 2 Emission inventory of airborne $PM_{2.5}$ in Prince George, Canada, for 2000 (based on data in Rubin et al. 2008)

Source	Contribution (%)
Point (mainly pulp mills, saw mills)	68
Road sediment	18
Area (mainly residential space heating)	13
Mobile (e.g. vehicle emissions)	1[a]

[a] Likely to be an underestimate of true value (Rubin et al. 2008)

Whilst the origins of sediment in agricultural and forested river basins have been studied extensively (e.g. Owens et al. 2000; Walling et al. 1999; Walling and Collins 2007), very limited study has been made of river sediment sources in urban basins. Carter et al. (2003) used the sediment fingerprinting approach to show that the sediment in the most urbanised sections of the Aire–Calder river basin in the UK was sourced from channel bank erosion (18–33%), uncultivated topsoil (4–7%), cultivated topsoil (20–45%), RDS (19–22%) and sewage input (14–18%). The high contribution of urban sources (up to 40% sewage and RDS) illustrates the marked contrast of urban sediments to those in non-urbanised river basins. Yin and Li (2008) used a similar approach using ^{7}Be and ^{210}Pb as tracers to estimate that about 60% of the suspended sediments at the outlet of a sewer system in Wuhan City, China was derived from the drainage system (gutter sediments and combined sewer sediments), with about 40% from RDS. These findings are in general agreement with Nelson and Booth (2002) who found that, as well as landslides and channel bank erosion, 15% of sediment in an urbanised catchment was from road surface erosion. Generally, there is a reduced *absolute* contribution of channel bank material in urban rivers, compared to non-urban rivers, due to the engineered nature of most urban channels and the protection of channel banks from erosion, and a reduced *relative* contribution of channel bank material due to sediment supply from additional urban sources such as RDS and sewage inputs. However, under certain circumstances, channel bank erosion can be a significant source of sediment in urbanised rivers, thus Trimble (1997) found that channel erosion (bank and bed) contributed about two thirds of the sediment yield from the urban area of San Diego Creek, California, USA.

At a national level, there are very few studies that estimate the contribution from urban sources to the sediment load of rivers. Table 3 gives values for England and Wales based on a recent study by Collins et al. (2007), which primarily used a combination of spatial databases and modelling. While highly generalised, the values shown in Table 3 demonstrate the important contributions from urban sources (both diffuse and point sources) to the total sediment loading of rivers at the national scale, especially given the relatively small (albeit growing) surface area of urban areas compared to agriculture. Further such studies are required before a full appreciation of the relative contribution of sediment (and associated contaminant) sources to urban rivers can be gained.

Table 3 Mean annual total sediment delivery to all rivers across England and Wales from various land uses (data from Collins et al. 2007; Collins and Anthony 2008)

Source of sediment	Sediment contribution	
	t	%
Diffuse from agricultural sector	1,929,000	75.7
Diffuse from eroding channel banks	394,000	15.5
Diffuse from urban sector	147,000	5.8
Point source (STWs)	76,000	3.0

In many cases, the sediment supplied to urban rivers is enriched in concentrations of metals and other pollutants. As well as contaminant input from road runoff and RDS, increased levels of nutrients, metals and micro-organic pollutants (e.g. pharmaceutical products) are sourced from STWs and CSOs (Gasperi et al. 2009; Meharg et al. 2003; Owens et al. 2001; Warren et al. 2003). Industrial processes also source metal contaminants to urban rivers (Walling et al. 2003), while upstream agricultural and forested areas are important sources of nutrients (e.g. P) and organic pollutants from fertilisers, herbicides and pesticides (Kronvang et al. 2003; Van der Perk et al. 2007) and some metals such as Cu from vineyards (Van der Perk and Jetten 2006). While many of the contaminants in urban rivers are delivered sorbed on sediment itself (i.e. from RDS, STWs and CSOs), some contaminants are delivered in solutional form and sorb onto sediment once in the channel network. For example, Owens and Walling (2002) describe how suspended sediment became enriched in P after passing a STW in the UK, with a large proportion of the P entering the river from the STW being in solutional form.

3.3 Sources to docks and canals

Compared to rivers, which receive sediment from a wide area, canal sediment is commonly dominated by material that is more locally derived, as a result of the limited transport of sediment in canals. This may be derived from industrial sources or sewage as well as natural material eroded from nearby land and road surfaces. However, major canal and dock systems which have significant water inputs from rivers can have a significant sediment source from outside the system. For example, Qu and Kelderman (2001) showed that sediment, and associated contaminants, in the Delft canals, the Netherlands, have been derived predominantly from the River Rhine, with the remainder coming from urban and industrial sources. Urban docks and canals also commonly receive high levels of organic matter, discharged from CSOs, and contaminants derived from boat traffic, for example hydrocarbons and tributyl tin (TBT) (e.g. Wetzel and Van Vleet 2003). Within urban lakes, sediment sources are generally a combination of both eroded soil material from the surrounding catchment and anthropogenic material from the urban environment. Atmospheric deposition may also be a *relatively* important source of particulates and associated contaminants, especially for lakes with no direct river input, and as such, provide good historic records of changes in atmospheric pollution levels (Charlesworth and Foster 1999).

4 Physical and chemical composition and characteristics of urban sediments

4.1 Road-deposited sediment (RDS)

RDS characteristically possesses high concentrations of metals (e.g. Charlesworth et al. 2003; Beckwith et al. 1986; Kim et al. 1998; Robertson et al. 2003; Robertson and Taylor 2007). Particular study has been made on the levels of Pb within urban street sediments, largely based on concern over human exposure to toxic levels of this contaminant. A significant enrichment in Pb levels in inner city centres has been documented (Duggan and Williams 1977; Carraz et al. 2006; Thornton et al. 1994; Robertson et al. 2003), supporting other evidence, particularly Pb-isotopic data (Zhu et al. 2001), that Pb is derived from petrol combustion. As explained above, more recent studies of Pb in RDS have shown a decrease in the levels of Pb, consistent with the reduced use of leaded petrol. For example, in 1975 average Pb levels were found to be 941 ppm in the city of Manchester, compared to a background level of 85 ppm (Nageotte and Day 1998). By 1997 levels had fallen to 569 ppm (Nageotte and Day 1998), and to 265 ppm by 2000 (Robertson et al. 2003). These relatively recent drops in sediment-Pb levels illustrate the transitory, short-term nature of these sediments within urban systems. For example, Allott et al. (1990), using radiocaesium from the dated Chernobyl fallout event, has documented the residence time of sediment on street surfaces to be short, in the order of 150 to 250 days.

Data on the chemical speciation of contaminants within RDS has provided information on the mineralogical affinity and potential reactivity of contaminants (Al-Chalabi and Hawker 1996; Stone and Marsalek 1996; Charlesworth and Lees 1999; Robertson et al. 2003; Fig. 3). In general, low levels are associated with the exchangeable (most easily solubilised) fraction (e.g. Charlesworth and Lees 1999). However, Cd and Zn have been observed to be associated with the exchangeable phase (Hamilton et al. 1984; Robertson et al. 2003) and, therefore, RDS may be a significant source of Cd and Zn to urban runoff. Platinum in urban sediments has been shown to be in a form that may be soluble (Farago et al. 1998) and street sediments in gully pots have also been shown to be actively mobilising Pt to the aquatic phase (Wei and Morrison 1994a). The majority of studies have found most metals to be associated primarily with the reducible (Fe and Mn oxides) fraction (e.g. Robertson et al. 2003; Stone and Marsalek 1996; see Fig. 3). Whilst this suggests that on street surfaces contaminant mobility is generally low, changes in pH and redox as a result of deposition in aquatic sediments or sediment–water transport would possibly release metals back into aquatic environments (see sections below). Copper has been shown to display a higher affinity to organic matter (Charlesworth and Lees 1999; Hamilton et al. 1984; Robertson et al. 2003; see Fig. 3). Much less direct information exists on the role of individual minerals on contaminant behaviour in urban sediments. McAlister et al. (2000) documented the stabilisation of weddellite (calcium oxalate dihydrate), derived from sewage, by interactions with metals in street sediments of Brazil. Serrano-Belles and Leharne (1997) documented the enhanced release of Pb from RDS upon the addition of chloride, in the form of salt, probably as a result of the formation of chloro-lead complexes. Recently, Taylor and Robertson (2009) documented the important role that iron oxides, derived from the oxidation of steel, and glass grains, derived from high temperature combustion processes, plays in hosting metals in RDS in Manchester, UK. RDS can also contain a high level of soluble anions and nutrients (e.g. nitrate, phosphate, chloride, ammonium, sulphate; Carraz et al. 2006), which highlights the potential impact upon surface water quality.

Physical study of RDS has shown it to possess a wide range in grain sizes, with some RDS being coarse in nature (Droppo et al. 1998, 2006; Robertson and Taylor 2007; Fig. 4), whilst some RDS is dominated by fine material <63 µm in size (Sutherland 2003). Particle size of the RDS may vary spatially within an urban environment, reflecting differences in sources and transport mechanisms. Thus, Droppo et al. (2006) documented finer sediment associated with industrial areas and coarser sediment associated with the commercial and residential area of an urban area adjacent to Hamilton Harbour, Ontario, Canada. The <63-µm-size fraction commonly exhibits the highest level of metals in RDS (Sutherland 2003; Droppo et al. 2006; Robertson and Taylor 2007; Table 4). For some contaminants, greater concentrations are associated with the coarser fractions. Thus, Sutherland et al. (2007) found that the PGEs were mainly in the >63 µm fraction for RDS samples collected from an urban basin in Hawaii, USA. On a mass-loading basis, the size fraction with the dominant contaminant loading will depend on: (1) the relationship between contaminant concentration and particle size; and (2) the relative abundances of the various size fractions within the RDS. Due to the coarse grain size of many RDS deposits, the coarse grain fractions often contain the most contaminant loading. In a similar manner to sediments from other environments (i.e. Horowitz 1991), the increase in contaminant loading in finer grain sizes is generally believed to be

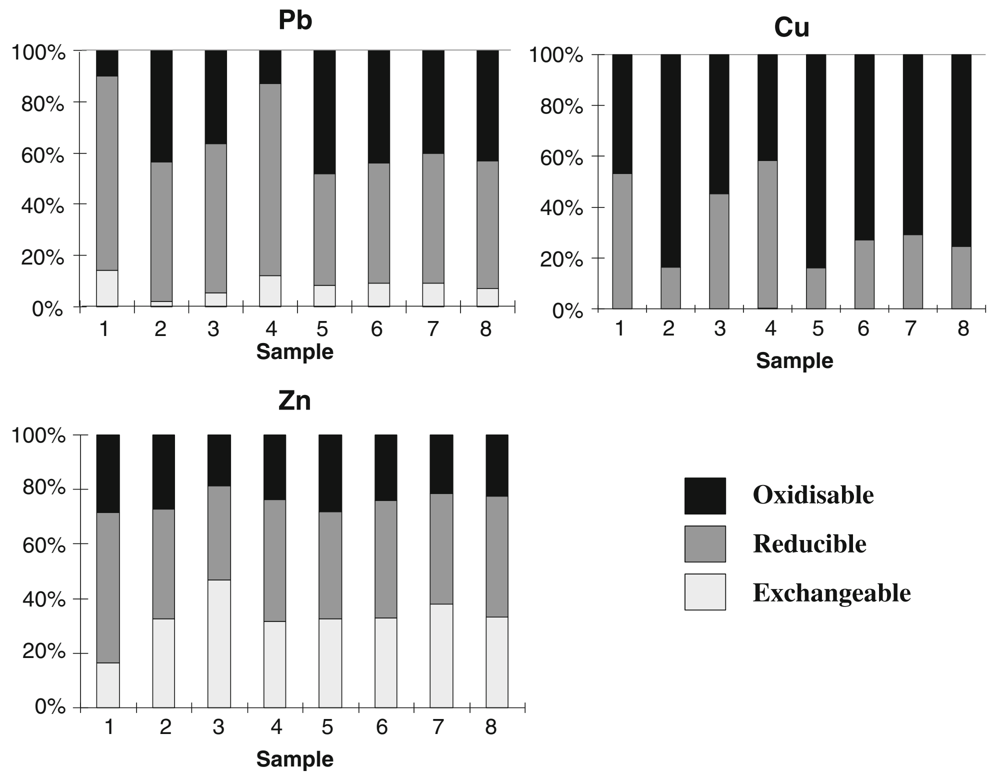

Fig. 3 Metal speciation within RDS from Manchester, UK, based on sequential extraction analysis (from Robertson et al. 2003; reproduced with the permission of Elsevier)

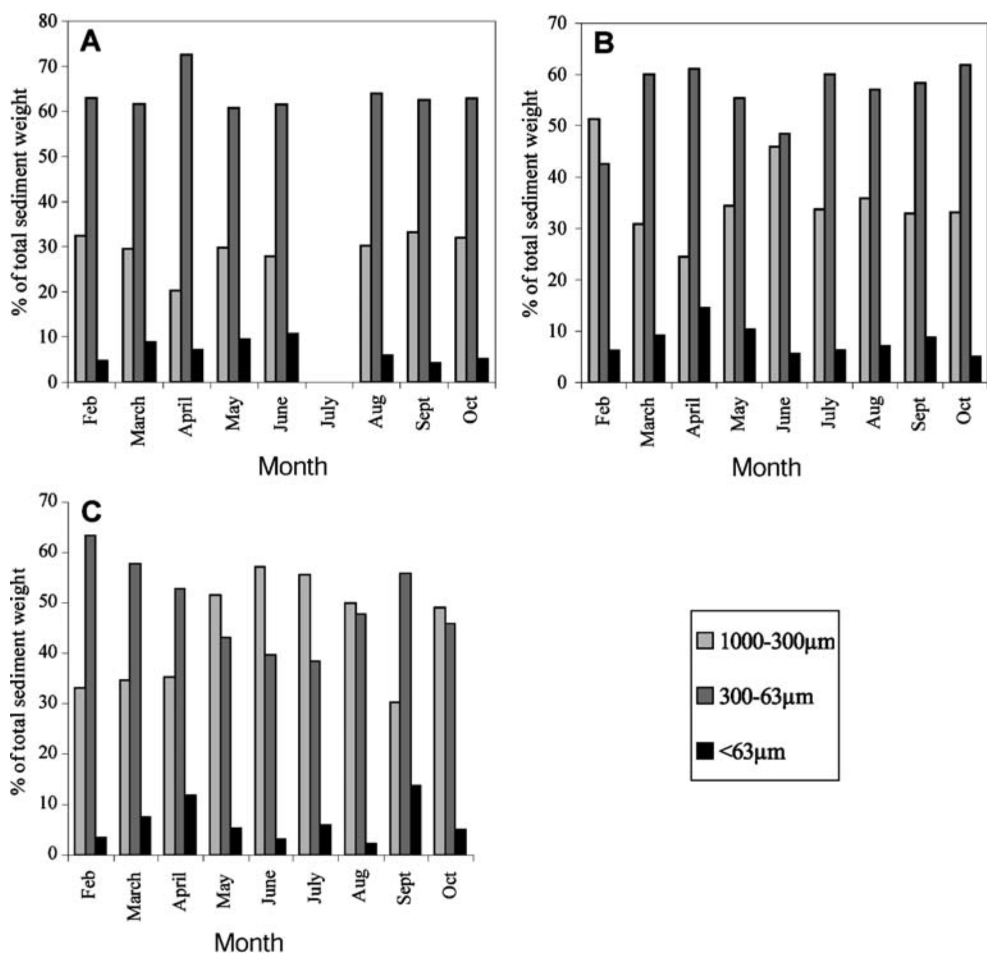

Fig. 4 Grain-size distributions, by weight, for RDS from Manchester, UK (from Robertson and Taylor 2007; reproduced with the permission of Springer)

Table 4 Grain-size-specific metal concentrations for RDS, Manchester, UK in μg g^{-1} (from Robertson and Taylor 2007)

	Pb			Mn			Zn			Fe			Cu		
	<63	63–300	>300	<63	63–300	>300	<63	63–300	>300	<63	63–300	>300	<63	63–300	>300
Feb	213	46	9	213	79	95	318	50	175	3,311	6,692	7,793	125	48	28
Mar	238	88	168	200	122	160	453	287	130	5,804	3,787	7,449	163	64	40
Apr	116	107	154	139	114	105	357	450	377	4,299	4,955	9,423	223	146	110
May	195	135	71	136	142	140	326	136	170	5,296	8,215	13,408	201	80	59
Jun	195	138	98	199	191	168	511	144	460	11,374	3,949	14,889	227	171	71
Jul	180	101	149	216	213	198	1,300	119	271	11,023	5,222	14,388	241	68	113
Aug	125	48	33	171	107	123	318	89	112	5,652	3,534	6,699	181	56	44
Sep	133	78	93	141	146	162	353	156	275	8,164	7,847	11,243	294	80	48
Oct	227	87	80	194	105	178	360	53	76	5,534	7,044	7,294	267	89	44

a result of the increased surface area with decreasing grain size, providing greater surface area for metal sorption to clay minerals or organic matter. In addition, the hydraulic selectivity of the sediment transport process by flowing water means that it is the finer particles (i.e. the <63 μm fraction) typically that are mobilised from the road network and exported towards the sewer system and river network (Droppo et al. 2006; Sutherland 2003). The recognition that contaminant loading and delivery are heterogeneously distributed relative to particle size is important when considering the management and pollution abatement of RDS.

Spatial variability in RDS contamination is also evident at a range of scales (e.g. Droppo et al. 2006). Studies on city-scale variability have shown that Pb levels are lower in outer city locations compared to inner city sites, indicating the role that traffic plays in the distribution of this contaminant (Carraz et al. 2006; Charlesworth et al. 2003; Duggan and Williams 1977; Massadeh and Snook 2002; Robertson et al. 2003; Fig. 5). Variability also exists across the street environment, with different levels of contaminants being present in gutter samples from those in street centres and pavements (Linton et al. 1980), highlighting the need to understand the appropriate scale for monitoring studies. There have been few studies on the temporal variability in RDS composition. A recent study in Manchester, UK (Robertson and Taylor 2007; Fig. 6) shows that significant monthly variability in composition exists at individual sites and this is most likely linked to seasonal weather patterns.

4.2 Gully pots and sewer systems

Gully pots are the first entry point of road runoff into the urban drainage network, and are designed to trap some of the sediment carried by the runoff. The design is usually one of a sump or a settling chamber, the entry of which is situated at the kerbside. They are a major feature of urban drainage networks, with more than 17 million present in the UK alone (Memon and Butler 2002a). The trapping of this sediment is desirable for two reasons. Firstly, it minimises the amount of sediment that enters the sewerage system and, thereby, reduces the problems caused by sediment accumulation in sewers. Secondly, where gully pots are emptied frequently, it minimises the amount of sediment that is potentially flushed out of the sewer system into rivers and receiving water bodies. The design and assessment of gully pots has been undertaken primarily in the field of civil engineering, where the term *sludge* is used to describe the sediment in a pot and *liquor* to refer to the in-place water.

The processes acting within gully pots are complex. During runoff events (*wet weather processes*), denser particles in the water will settle out under gravity. However, there is usually a high degree of turbulence within the gully pot, which not only limits the amount of sediment that will settle out but may also lead to the erosion and resuspension of existing sediment in the pot. Biochemical changes also take place within the gully pot. Most biochemical changes take place during periods between runoff events (*dry weather processes*). Chemical studies of gully pot liquor have shown that water reaches negative Eh during dry period events, with major impacts upon redox-sensitive chemical species. The major changes are the consumption of oxygen in liquor, changes in chemical oxygen demand (COD) and ammonium, and the release of contaminants and nutrients adsorbed onto redox-sensitive species into the pot liquor (Wei and Morrison 1994b; Memon and Butler 2002b). Dry weather processes overall increase the pollutant level of pot liquor, with subsequent negative impacts on

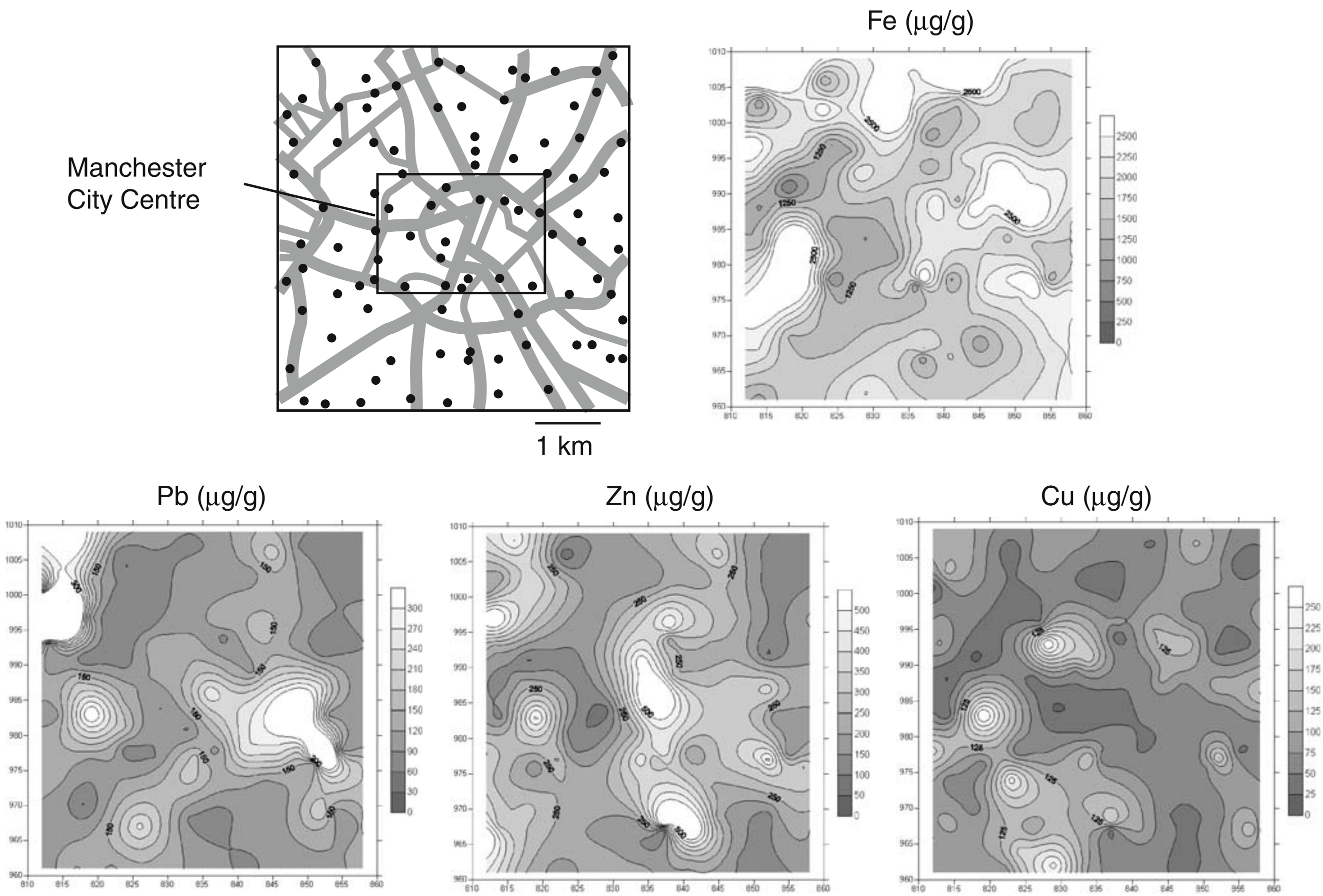

Fig. 5 Spatial distribution of metals and magnetic susceptibility in RDS in the city of Manchester, UK (from Carraz et al. 2006; reproduced with the permission of Manchester Geographical Association)

runoff into waterways during wet weather processes. Thus, the longer the period of dry weather then the greater these changes are likely to be (Memon and Butler 2002b).

4.3 Urban rivers

4.3.1 Sediment–contaminant impacts

During the process of urban development, rivers are commonly modified to enable navigation, culverted to minimise flooding, and diverted to allow development. These changes lead to the physical modification of rivers, to the extent that urban rivers commonly possess unique physical and hydrological properties (Gurnell et al. 2007). In addition to this physical modification, urban rivers historically became the major vectors of domestic and industrial waste removal. A direct result of this is high pollution levels through both organic and inorganic contamination. It is only with increased discharge legislation and investment in sewer networks and treatment plants that urban rivers are now improving in water quality in developed nations. However, the majority of the contaminant load in rivers is associated with the sediment fraction (Horowitz 2008; Savenko 2006; Viers et al. 2009). The residence time of sediment in rivers is greater than that of water, and in spite of water quality improvement many urban rivers still possess low sediment quality. This legacy of pollution is one of the largest problems facing urban catchments.

Sediment within urban rivers is considered itself to be a major non-point source pollutant, as fine sediment causes a range of problems in rivers (Owens et al. 2005; Bilotta and Brazier 2008). An increase in fine sediment has impacts upon river turbidity, impacting biological processes and ecology. Thus, Walters et al. (2003) describe how sediment delivery and deposition resulted in homogenised fish assemblages in the urbanised Etowah river basin, USA, by creating turbid streams and channel beds containing finer sediments. The impacts of high suspended sediment concentrations upon fish are marked and include effects on behaviour and health of fish through gill damage and damage to spawning grounds, through infilling of spaces between sands and gravels, and reducing oxygen delivery to the fish eggs (Watts et al. 2003). These changes in

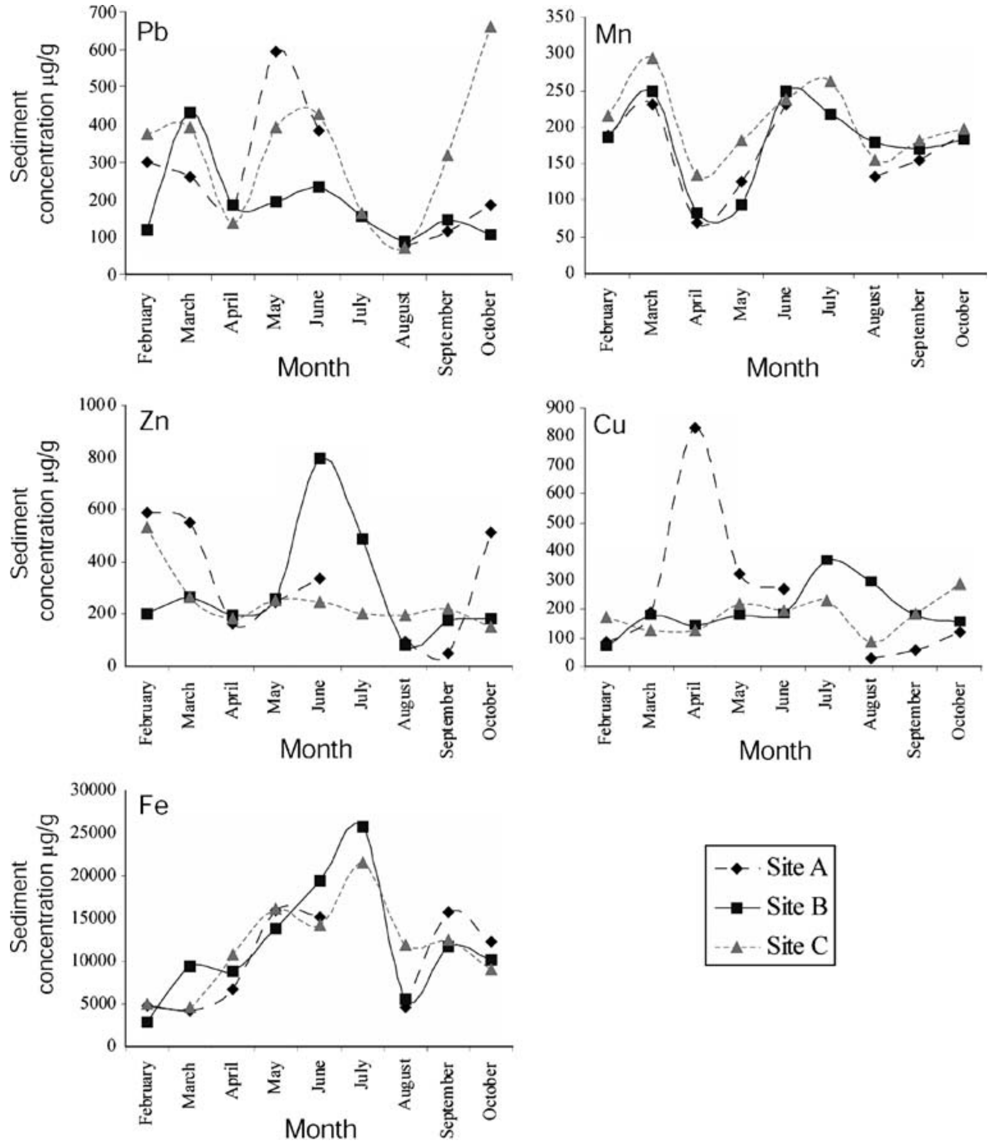

Fig. 6 Monthly variability in metal contents of RDS from three sites in Manchester, UK (from Robertson and Taylor 2007; reproduced with the permission of Springer)

sediment supply and sediment quality (composition and particle size), illustrated above, can have major impacts upon ecology and urban biodiversity of rivers, which in turn greatly influence river ecosystem services.

Changes in sediment delivery (both increases and decreases) coupled with changes in river flows due to urbanisation can have a physical impact on channel morphology (Chin 2006), often causing channel enlargement (Gurnell et al. 2007). In some cases, increased sediment delivery causes channel aggradation which may lead to channel volume reductions and an increased risk of flooding. Urbanisation may also cause changes in channel bed sediment texture (Sciera et al 2008). Thus, Finkenbine et al. (2000) reported the selective removal of fine sediment from the channel beds of urban rivers in Vancouver, British Columbia, Canada.

4.3.2 Sediment and contaminant concentrations and fluxes

Due to both the impervious nature of urban land surfaces and the engineered nature of urban rivers, the flow in such rivers responds rapidly to rainfall events. This leads to a rapid rise and fall in the river level, and the river is said to display a flashy response to rainfall. This leads to high rates of fine sediment transport in suspension during these flow events, often several orders of magnitudes more that during low flow periods. For example, Horowitz et al. (2008) found that >94% of the transport of suspended sediment in the City of Atlanta, USA, occurred during storm events which occupied <20% of the time. Similarly, Old et al. (2003) report that 40% of the annual sediment load of the urbanised Bradford Beck, UK, was transported in about 1% of the time due to high inputs from CSOs during flow events. Interestingly, Old et al. (2003) also found that proportionally more sediment was transported in the summer season in urban areas compared to non-urban parts of the basin, reflecting the fact that urban areas are able to generate runoff more easily than non-urban, rural areas in the summer months.

Urban systems may exhibit a ‘first-flush’ or ‘initial micro-pulse’ (Lee et al. 2002; Lawler et al. 2006; Yin and Li 2008) whereby there is an initial increase in the sediment flux (concentration and/or load) due to a coupling between urban sources (i.e. RDS, CSO delivery) and the urban river network by rain events. First-flush effects tend to weaken with increasing river basin area (Lee et al. 2002). Typically, sediment fluxes decrease through time due to sediment exhaustion effects, either during a prolonged storm event or during successive storms in quick duration; i.e. fluxes get smaller with each storm due to decreasing availability of easily mobilised and transported sediment. Some studies (e.g. Lawler et al. 2006), however, have shown that fluxes may continue after the peak in discharge due to: (1) additional sources not directly linked to discharge, such as CSOs; (2) in-channel processes such as biofilms increasing the resistance for mobilisation of channel bed sediment until the biofilms are degraded; and (3) the arrival of sediment from distal sources, such as distal road sources and the upstream non-urbanised part of the basin which may have longer transit times.

Suspended sediment concentrations (SSC) in urban rivers can be relatively high. Gromaire-Mertz et al. (1999) compiled data showing the mean SSC to be in the range 49 to 498 mg l^{-1} for high-flow events. Gromaire et al. (2001) also reported sewer outlet SSCs in the range 152 to 670 mg l^{-1}, illustrating the importance of sewer outfalls in urban river basins. A study of the Bradford Beck, UK (Old et al. 2003) documented high levels of suspended sediment transport during a single storm event. Suspended sediment concentrations increased from 14 to 1,360 mg l^{-1} over a period of 15 min. A peak sediment flux of 47.2 kg s^{-1} was recorded, illustrating the high levels of sediment that are transported by urban rivers during high flow, and that it is these short-lived events that dominate sediment movement.

Suspended sediment yields from rivers in urban basins tend to fall within the range 40 to 500 t km^{-2} $year^{-1}$ (Goodwin et al. 2003; Old et al. 2006; Wanielista et al. 1997), although they may be higher in some situations (Horowitz et al. 2008), especially during the initial stages of urban development and construction (Wolman 1967). This range is similar to values documented for agricultural basins, although are lower than those reported for areas undergoing severe disturbance (i.e. active deforestation, wildfires etc.), and for basins draining high mountains undergoing active tectonics (Caine 2004), primarily reflecting differences in sediment supply rather than transport capacity.

Studies of urbanised rivers have shown the clear increased concentrations and fluxes of contaminants associated with the fine suspended sediment fraction, compared to the coarse or soluble fractions. In the City of Atlanta, USA, Horowitz et al. (2008) found that >75% of the annual fluxes of major and trace elements and P occurred in association with suspended sediment. Interestingly, although concentrations of sediment-associated chemical constituents in rivers in Atlanta tended to decrease during storm events, fluxes tended to increase reflecting increased mobilisation and transport of suspended sediment. In the case of the Rivers Aire and Calder, UK, Owens et al. (2001) and Walling et al. (2003) showed that concentrations of metals (Cr, Cu, Pb, Zn) and PCBs on sediment increased significantly in urbanised sections of these rivers. Thus, Cr in suspended sediment increased from around 100 μg g^{-1} in non-urban sections to around 400 μg g^{-1} in urbanised sections, and PCBs showed a similar four-fold increase. Similarly, Owens and Walling (2002) documented a clear increase in sediment-bound P (a major contributor to river eutrophication) as a result of urbanisation in the same rivers. They documented changes in total P from average values of <2,000 μg g^{-1} in upstream sections to over 7,000 μg g^{-1} in sections downstream of urbanisation. They concluded that this increase represented point inputs of P from STWs and CSOs. The input of these point sources was further supported by a change in the inorganic P to organic P ratio from <2 upstream to >4 downstream of urban centres. This is significant in that inorganic P is more bioavailable than organic P.

Studies have also documented the speciation of contaminants associated with sediment, especially metals, in urban rivers. Thus, Carter et al. (2006) report that metals (Cr, Cu and Pb) were mainly associated with Fe and Mn oxides and organic material for suspended sediment samples collected from the River Aire, UK (Fig. 7), although they found important spatial and temporal variations, and that speciation was metal dependent. Other studies (e.g. Macklin and Dowsett 1989; Pardo et al. 1990; Salomons and Förstner 1984; Tessier et al. 1979, 1980) have also identified the importance of the Fe and Mn oxides and organic fractions for metal sorption. Carter et al. (2006) also reported that the organic content of the suspended sediment in urban reaches is higher than that for sediment in upstream rural locations, reflecting inputs of organic-rich material from CSOs and STWs.

An important feature reported in several urbanised river systems is the degree and role of particle flocculation. While not unique to urban systems, the enhanced supply of flocculated material from STWs and CSOs (Droppo et al. 2005), in addition to the increased delivery of organic

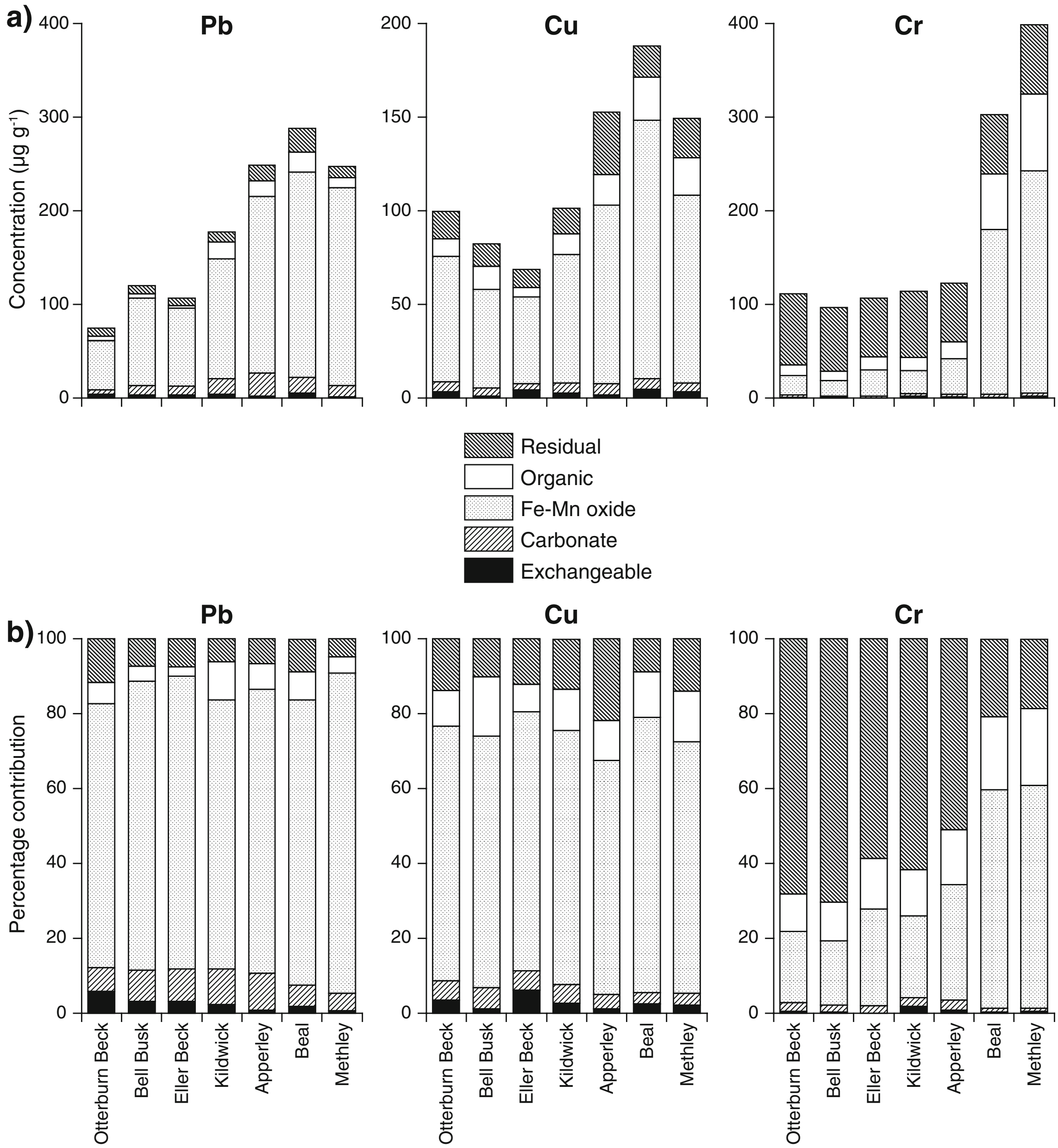

Fig. 7 Downstream change in (**a**) concentration and (**b**) the proportion of metals associated with each chemical fraction from seven sites in the basin of the River Aire, UK (from Carter et al. 2006; reproduced with the permission of Wiley-Blackwell)

material from these sources (which increase the likelihood of flocculation), means that flocculation processes are likely to be more important within urban rivers. Flocculation processes influence the structure and size of sediment particles (Droppo 2001). Importantly, the creation of such 'flocs' influences their hydro-dynamic properties, in particular their settling velocity, and the sorption capacity of the composite sediments, both of which influence contaminant transport and storage (Droppo et al. 1998). Several workers have documented that point source inputs influence the degree of flocculation which, in turn, can influence the amount and particle size composition of channel stored sediments and associated contaminants. Thus, Petticrew and Biickert (1998) found that channel bed sediments in the Fraser River downstream of a pulp mill effluent outlet in the city of Prince George, Canada, were enhanced in fines relative to upstream samples reflecting deposition of flocculated particles.

4.3.3 Storage of sediments and contaminants

Generally, the storage of sediments and associated contaminants in urban river channels occurs in two main features: on the channel bed and on floodplains. For channel bed storage, Lawler et al. (2006) report values of fine-grained sediment deposition of ca. 50–100 g m^{-2} for the urbanised Upper River Tame, UK, and Walling et al. (2003) found values within the range 100–1,450 g m^{-2} for the Rivers Aire and Calder, UK. The latter study also estimated annual mean storage for a range of contaminants, with values up to ca. 0.5 g m^{-2} for Cr and Pb, 1.3 g m^{-2} for Cr, 6.3 g m^{-2} for total P and 0.13 mg m^{-2} for total PCBs for the downstream site on the River Calder.

In addition to the deposition of sediment particles, often as flocs, onto the channel bed, studies (Gainswin et al. 2006a, b; Jarvie et al. 2005; 2006) have shown that channel bed sediments may act as important sinks (and, at times, sources) for soluble pollutants, such as soluble reactive P, in overlying water. In addition to concentration gradients between overlying waters and channel bed sediments (Jarvie et al. 2005), particle size and the presence of biofilms on larger sediment particles (gravels and stones) are important in controlling sorption–desorption rates (Gainswin et al. 2006a, b). Furthermore, in-stream biota such as macrophytes, periphyton and phytoplankton are important for the retention and cycling of contaminants, such as P, through interception, trapping, sorption and uptake processes (Withers and Jarvie 2008).

While locally important, and significant for short durations, generally channel bed storage is a relative small portion of the overall sediment and contaminant budget of urbanised river basins. At the river basin scale, Walling et al. (2003) determined that channel bed storage typically represented < 3% of the annual flux of sediment-associated contaminants (metals, PCBs, P) measured at the downstream gauging station on the River Aire (i.e. below the urbanised area) (Table 5). This finding partly reflects the mid- and downstream locations of the urban area within this mid-sized (ca. 2,000 km^2) river basin. In basins where urban areas are located in upstream areas and/or where the urban area occupies most of the basin, then channel bed storage of contaminants will likely assume greater importance.

At a national level, Horowitz and Stephens (2008) document the importance of urban sources in influencing the geochemical content of channel bed sediment in the conterminous USA. They found that sediment samples collected from urban areas (as identified by metrics such as population density and urban percentage) were enriched in sediment-associated contaminants, such as metals, P and C, relative to baseline values and for basins impacted by agricultural and forest land uses. As such, the contaminant content of channel bed sediment represents a useful method to identify the influence of urban and other activities on pollution levels in rivers.

The other main storage feature within the urban river system is floodplains. However, river floodplains tend to occupy less area than in rural and agricultural areas due to the presence of urban development on floodplains and, consequently, the protection of such land from flooding. Where floodplains do occur, then rates of overbank sedimentation can be high, resulting in locally important stores of sediment and contaminants. Walling et al. (2003) estimate that the annual conveyance loss of sediment-associated contaminants to floodplain storage for the River Aire basin of between 2% and 26% depending on contaminant (see Table 5). Generally, contaminants are locked within the sedimentary profile, although there may be some limited diffusion and migration depending on bioturbation, flooding cycles, water table fluctuations, break-down of organic materials, and changing pH and redox conditions (Hudson-Edwards et al. 1998). Thus, downcore profiles of contaminant levels in floodplain sediments offer the potential to reconstruct past changes in sediment contaminant levels in urban river basins (e.g. Owens and Walling 2003). Indeed, a recent initiative (Bølviken et al. 2007) is using the difference between the contaminant content of surface and sub-surface floodplain

Table 5 Estimates of the mean total channel bed storage of contaminants, the average annual total deposition flux of contaminants to floodplains, the total suspended sediment-associated loads at the catchment outlet and the conveyance losses of contaminants associated with overbank floodplain deposition for the River Aire, UK, during the period December 1997 to December 1999

Contaminant	Mean total channel bed storage (kg)	Mean annual floodplain deposition (kg $year^{-1}$)	Mean annual suspended sediment-associated load (kg $year^{-1}$)	Conveyance loss (%)
Cr	35.0 (0.4)	250 (10)	2,509	9.1
Cu	78.5 (2.8)	376 (14)	2,761	12.0
Pb	97.4 (2.7)	1,302 (36)	3,658	26.3
Zn	229 (3.0)	2,428 (24)	9,988	19.6
Total P	1,468 (1.2)	11,479 (10)	120,205	8.7
Total PCBs	0.0024 (0.04)	0.161 (2)	6.5	2.4

The values in parentheses are the estimates of storage and deposition flux expressed as a percentage of the annual load at the catchment outlet (modified from Walling et al. 2003) sediment to identify the influence of anthropogenic activities, including urbanisation, on sediments transported by various rivers throughout the world.

4.4 Urban canals and docks

The major sites for sediment accumulation in large urban environments are canals, docks and lakes, and these are often the terminal receiving water bodies in urban river basins. Canals and docks contrast with lakes in being artificial and heavily engineered so possess unique hydrological and sedimentological properties. Transport, especially for industry, was a major component in urban areas and as such navigable waterways and docks were built to accommodate this. These are most commonly freshwater in nature, but docks in coastal urban environments may be estuarine or marine, and in the case of Venice, Italy, canals may also be marine in nature. Due to the low-flow conditions in canals and docks, and the steep-sided nature of these water bodies, they are highly depositional in nature, resulting in the rapid accumulation of sediment. This leads to the requirement for regular dredging to preserve navigable status.

Sediments deposited within urban canals and docks are predominantly derived from anthropogenic sources, and these anthropogenic sediments have mineralogical and geochemical compositions which contrast significantly to those of natural sediments. However, to date, very little research has been directed towards detailing the mineralogy and geochemistry of these sediments. Two aspects of urban canal and dock sediments are apparent: they commonly have a high organic matter loading as a result of historical sewage input (Taylor et al. 2003) and have high metal loadings as a consequence of industrial waste and discharges (Bromhead and Beckwith 1994; Kelderman et al. 2000; Dodd et al. 2003). This high organic and contaminant loading may lead to significant impacts upon overlying water quality as a result of post-depositional chemical alteration.

Early diagenetic reactions also impact upon the short- and long-term fate of contaminants in sediments in docks and canals through two principal mechanisms: release of contaminants into sediment porewaters; and the uptake of contaminants into authigenic mineral precipitates. Within contaminated sediments, metals commonly co-precipitate with Fe and Mn oxides. The chemical reduction of these oxides results in the release of these adsorbed contaminants to sediment porewaters (Dodd et al. 2003; Taylor et al. 2003) and thence to overlying water. The build-up of chemical species in sediment porewaters also leads to the precipitation of authigenic minerals in the sediment. Within marine and brackish sediments the predominant mineral formed in this way is pyrite (FeS_2). Pyrite has been observed in canal sediments (Large et al. 2002; Taylor et al. 2003) but the absence of sulphate in freshwater leads to this being a rare mineral in urban sediments. The limited studies of the diagenesis of urban sediments have shown the iron phosphate mineral vivianite ($Fe_3(PO_4)_2{\cdot}8H_2O$) to be the most common mineral (Dodd et al. 2003, Taylor and Boult 2007). The importance of these minerals for contaminant mobility is that metals can be taken up by these minerals as they precipitate, thereby locking up contaminants in the sediment (Large et al. 2002; Taylor et al. 2008b).

5 Management of sediments in urbanised river basins

As is clear from the discussion above, the impacts of sediment particulates in urban river basins are wide ranging and include: impacts upon water quality and aquatic ecosystems; reduction of capacity in drainage systems; and a reduction in depth of navigable waterways. As a consequence, the active management of sediments is a key requirement in the sustainable urban environment. Many of the management activities described below are, or could be, part of sustainable urban drainage systems (SUDS).

5.1 Road-deposited sediment, gully pots and stormwater ponds

The routine physical removal of RDS is carried out in many urban environments, both for aesthetic reasons and to limit the impact of RDS on watercourses and sewer systems. Removal is accomplished by mechanical street sweeping. Early street-sweeping procedures were carried out predominantly for aesthetic reasons, and removal efficiencies were low (Sartor et al. 1974). Subsequent studies have shown that regular removal of RDS by street sweeping may lead to significant reduction in both sediment contamination levels and contamination of surface runoff (e.g. Sartor and Gaboury 1984). It can, therefore, be a highly effective method of urban pollution management. It has been shown that street sweeping is most effective at removing pollutants in climates where long periods of dry weather lead to pollutant accumulation (Sartor and Gaboury 1984). Street sweepers can be based on a vacuum system or on a rotary brush system, and comparisons have been made on the effectiveness of each type for RDS removal. Generally, whilst the rotary type removes a greater proportion of RDS from street surfaces, vacuum-based models are better at removing the finer grain fractions (Brinkmann and Tobin 2003). There is, therefore, a trade-off between an increased volume of sediment removal or more efficient contaminant removal. In general, street sweeping is much less efficient at removing the finer-grained fractions of RDS than the coarser-grained fractions. This has implications for pollution management as the finer-grained fraction generally contains the highest loading of contaminants. Consequently, several studies (e.g. Pitt and Clark 2003) have shown that street-sweeping methods may be relatively ineffective in removing contaminants associated with finer RDS. In a study in Hawaii, Sutherland (2003) showed that street sweepers only removed 62% of Pb from RDS, primarily as a consequence of low efficiencies of fine-grained sediment removal (Table 6). The resulting waste produced from street sweeping can be re-used as ground cover or disposed of on land or to landfill, but this waste has not been widely assessed for its suitability for such. The limited studies that have been undertaken (e.g. Viklander 1998; Clark et al. 2000; German and Svensson 2002) have concluded that the sweepings material, due to high contamination levels, should be treated prior to its re-use or disposal on land.

The management of gully pots, and their associated sediment, forms an important part of urban water quality management. Most authorities regularly empty and clean gully pots, thereby removing the sediment from the urban drainage system. However, this is commonly carried out to minimise flooding and drainage issues, rather than for pollution management reasons. Memon and Butler (2002a) modelled the efficiency of gully pots in urban drainage networks and showed that gully pots can reduce the suspended sediment content of water entering sewer systems (and ultimately receiving water bodies) by 40%, with even larger reductions being possible with improved gully pot design. However, the model also showed that reduction in pollutants (such as ammonium and chemical oxygen demand) were minimal.

Stormwater ponds are designed and engineered to remove pollutants from stormwater runoff, primarily through the settling out of sediments from the water column. As the majority of the pollutant loading is associated with the sediment, this leads to an improvement in the quality of the stormwater runoff, which can then be discharged to natural water bodies. Pollutant removal efficiencies of up to 90% have been reported for such ponds (e.g. Wu et al. 1996). Removal of P has also been documented, but the release of N in the form of ammonia from the sediments to the water column may also take place (Hvitved-Jacobsen et al. 1984). The build-up of sediments in these ponds leads to a reduction in volume and, therefore, efficiency. Therefore, sediment needs to be removed periodically, although these sediments are commonly contaminated. For example, Marsalek and Marsalek (1997) determined for a stormwater pond in Ontario, Canada, that sediment metal levels were such that the sediment could not be re-used or placed in residential landfill without treatment.

Table 6 Estimation of RDS Pb load removal per grain size fraction by street sweepers of varying efficiencies for Palolo Valley, Oahu, Hawaii (from Sutherland 2003)

Particle size fraction (μm)	Mean Pb loading (%)	Mean street sweeper efficiency (%)	Pb load removal (%)
1,000–2,000	2.7 ± 1.6	83.7	2.2
500–1,000	10.8 ± 6.2	81.7	8.8
250–500	13.4 ± 6.2	79.3	10.6
125–250	12.8 ± 6.3	75	9.6
63–125	9.2 ± 2.4	66.7	6.5
<63	50.6 ± 14.9	48.7	24.7
Total			62.4

Notes: mean sweeper efficiencies based on three sweeper types (mechanical, regenerative air and vacuum-assisted dry sweeping) and values are likely to be overestimates

5.2 Urban river system

Within the urban river network, management tends to consist of two approaches. One is to attempt to control, and ideally reduce, the delivery of sediment and contaminants. Reducing the delivery of contaminants from point sources, in particular, has received considerable attention and effort in recent decades. There is evidence (e.g. Owens and Walling 2003) to suggest that this has been effective in cities in developed countries, partly due to legislation and initiatives aimed at improving water quality (Casper 2008). In developing countries, the trend is less clear. The second main management approach is to remove sediments within urban river channels, canals and docks via dredging (maintenance dredging). This approach can be costly, especially if the sediment is contaminated, in which case typical options include environmental dredging, capping and monitored natural recovery. For example, the city of Hamburg, Germany, has to dredge between 2 and $5\times10^6 m^3$ of sediment each year from the River Elbe to maintain its port. It is estimated that the costs of this are of the order of 30 million euros annually, not including personnel and capital costs (Netzband et al. 2002). Similarly, Zeller and Cushing (2006) describe that the cost of dealing with contaminated sediment as part of 71 major environmental remediation projects in the USA, mainly since 1990, amounted to nearly US$ 1 billion.

While these two approaches are undoubtedly important in managing the amount and quality of sediments in urban rivers, generally they are undertaken without full appreciation that the urban environment is positioned within a larger landscape. As such, urban rivers receive water, sediment and chemicals from areas outside of the immediate urban area: typically upstream forested, rural and agricultural areas (see Fig 2). For management to be effective, this recognition is important. Sediment management within the context of the river basin represents a suitable concept (Owens 2005, 2008). However, urban river systems also receive sediment and contaminants from outside of the river basin (e.g. atmospheric deposition, see Fig. 2), and management of these sources may require different approaches.

5.3 Remedial measures for urban sediments: ecological risk-based assessment and sediment stability assessment

Several recent developments in the field of contaminated sediment offer considerable promise for sediment management within urbanised systems. One such development has been risk assessment for prioritisation of remedial measures. The concept of environmental risk assessment has a long history, especially in North America, associated with the management (e.g. dredging) of contaminated sediment, amongst other things. The approach has developed into one of ecological risk assessment (Suter 2008) and now lends itself to more integrated basin-scale and regional assessment and management (Förstner and Apitz 2007; Suter 2008). Heise and Förstner (2006) describe a three-step approach in order to prioritise sites in the river basin with regard to the risk that they pose for downstream areas. The approach requires:

- identification of the areas of substances of concern and classification of these into 'hazard classes of compounds';
- identification of areas of concern and their classification into 'hazards classes of sites'; and
- identification of areas of risk.

Figure 8 illustrates how this approach has been used to identify and classify areas of risk based on historically contaminated sediment in the Rhine basin in Germany. The risk assessment approach allows for prioritisation of remediation actions based on a sense of urgency. The concept also fits in well with WFD philosophy where priority substances are identified and remediation measures developed via the POM approach. Such an approach offers considerable potential for urbanised river basins given the emphasis placed on historical (contaminated) sediments and in identifying sources (substances/contaminants and areas/sources) of risk. The risk approach described in Heise and Förstner (2006), which mainly considers risk within the river channel network, could be extended to areas outside of the immediate channel network and thus could be used to prioritise areas and actions in other parts of the river basin (e.g. Fig. 2) such as RDS, gully pots and stormwater ponds. This approach could also include ecological risk-based assessment of water and air within river basins. This would lead to a more integrated risk-based decision framework for natural resources, which is in keeping with European environmental policy, such as the WFD (Apitz 2008).

Recent work has also considered sediment stability assessment, often as part of a risk-based assessment approach (Bohlen and Erickson 2006). Sediment stability analysis is particularly important in urban river channels because: (1) increases in the amount of impervious surfaces within the urban environment can increase the rate at which water is delivered to, and moves through, the urban river system, which in turn can increase shear stresses on bed sediment; and (2) because in many older urban river systems it is often the older, deeper sediment layers which are more contaminated, and thus there is a concern of remobilising 'legacy sediment'. Indeed, the remobilisation of such contaminated 'legacy sediments' during high-magnitude flood events has received considerable management attention (Bohlen and Erickson 2006) and research interest (e.g. Förstner et al. 2004; Westrich and Förstner 2007; Wölz et al. 2009), particularly given concerns over increased high-flow events associated with global environmental changes.

6 The impact of local and global environmental changes

The discussion above has been based mainly on our present knowledge and understanding, albeit imperfect, of sediment–contaminant sources and dynamics in urban systems. While we do not know what the future holds for urban systems, we do know that the urban environment is a dynamic one and will be subject to pronounced changes in the decades to come due to the impact of local, regional and global changes. The most important of these is a trend for an increase in the number of people in urban areas and the reliance of people outside of such areas for services (materials, goods etc.) which are produced within urban centres. Such growth in population and industry is likely to mean an increase in sediment and contaminant loadings in urban rivers and increased pressure on water resources. While the cause of such growth may be locally or regionally driven, particularly in developing countries, the overall trend is a global one. The role of sediment as a source and a vector for contaminant transfers within urbanised river basins is, therefore, likely to increase.

The trend for an increase in the proportion of people living in urban areas, coincides with, and perhaps is a symptom of, global climate change. Such climate changes vary from region to region, but tend to include increases or decreases in precipitation and increases or decreases in temperature (Kundzewicz et al. 2008). Kataoka et al. (2009) show how temperatures in seven large Asian cities have increased by an average of 2.5°C during the twentieth

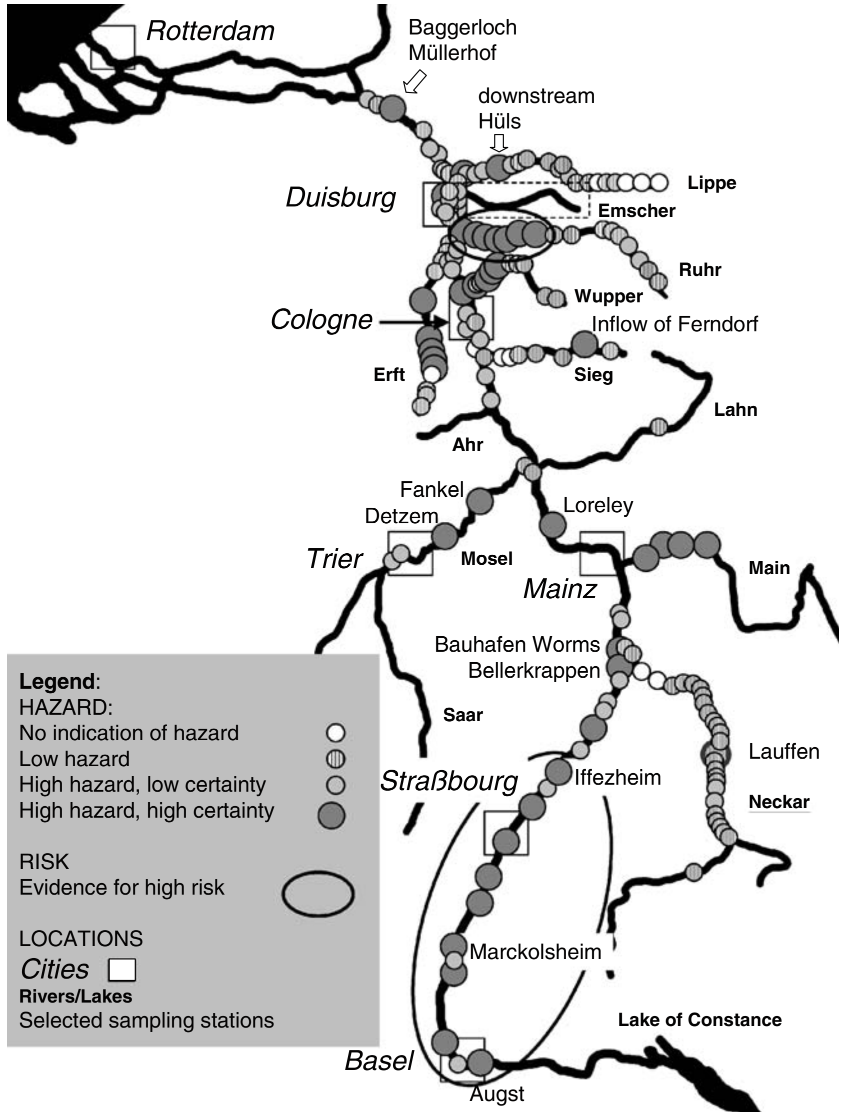

Fig. 8 Sediment–contaminant risk assessment for the Rhine basin, based on the need to mitigate downstream sediment issues in Rotterdam Port (from Heise and Förstner 2006; reproduced with the permission of Springer)

century, due to both urban warming and global warming, which compares with a mean world increase of about 0.7°C for the same period.

Whilst numerous studies have been undertaken on the impacts of climate change on natural systems, little consideration has been given to engineered environments. However, changes in climate may alter rainfall and snowmelt patterns, thereby impacting on drainage and flow regimes in urban rivers. Semadeni-Davies (2004) modelled the impact of possible future climate change on a cold region and city and showed that frequency and volume of wastewater flows in an urban environment would be altered, with implications for drainage system design and management. Similarly, Vergara et al. (2007) document how rapid glacier retreat in the tropical Andes of South America will decrease the availability of water resources for large cities. Climate change is also likely to influence the temperature of water in the urban landscape which will likely have implications for reactions between sediments and contaminants, in addition to direct impacts on aquatic habitats. Thus, while it is recognised that regional and global environmental changes will greatly impact urbanised areas, at present it is unclear how such changes will influence sediments within urban river basins.

7 Conclusions and recommendations

This review has demonstrated that sediment and associated contaminant dynamics in urban systems are complex. Sediments and contaminants are derived from a variety of sources, which can be from:

- inside the urban area (e.g. RDS, STWs, CSOs, industrial point sources)

- outside the urban area but within the river basin (e.g. upstream agricultural and forested areas, downstream estuarine areas)
- outside the river basin (e.g. allochthonous dust and airborne chemicals).

As such, urban sediment systems are ‘open’ (Chorley 1962) sub-systems within a wider landscape. Despite the fact that some sediment and contaminant sources are derived from outside the river basin, it still seems appropriate that urban sediment dynamics and management are assessed at the river basin scale. However, there is clearly a need to identify all sources and fluxes. As with any part of the river basin, urban systems are themselves influenced by other parts of the basin, especially upstream areas, and in turn urban systems influence downstream areas, particularly through the fast response of water and sediment flows to rainfall and through the delivery of contaminated sediments. Consequently, urban systems are likely to be one of the dominant influences on the water quality and aquatic ecology of most forms of water (surface water, groundwaters, estuarine and near-coastal waters) in river basins. As such, they merit greater attention than has been given in recent years, particularly given anticipated increases in urban populations and surface areas.

Urban systems are an important source of sediment and chemicals to the global oceans. While studies have investigated riverine fluxes of sediment and chemicals to the global oceans (Savenko 2006; Walling 2006; Viers et al. 2009), attention has mainly focused on the role of deforestation, agriculture, soil conservation, dams etc., and the precise role of urban areas in controlling these fluxes has yet to be addressed adequately.

We conclude by listing some recommendations for future research: (1) There is a need for better sampling and monitoring of sediment and sediment-associated contaminant fluxes and cycling in urban river channels and basins, particularly recognising the complex interactions between sediments, chemicals and water in such environments (cf. Horowitz 2008). This should include better techniques and studies to identify sources and transfers of RDS, airborne particulate matter and sediments in the river system. (2) Greater interdisciplinary research is needed, combining sedimentologists, hydrologists, chemists, biologists, urban planners, urban and industrial archaeologists, and social and economic scientists. While environmental and physical scientists can provide the necessary information to understand how the urban system behaves, social and economic scientists and policy makers are needed to turn this into meaningful planning and management actions. (3) More attention needs to be focussed on upscaling, and connecting urban areas to the rest of the river basin, both upstream and downstream. (4) Finally, there is a need to balance multiple needs (urban population, water resources) with likely trends in urban development and global environmental change.

In a recent review of trends and developments in sediment–water science over the last 30 years, Petticrew (2009) has identified the increasing use of science for improved management of aquatic systems. It is hoped that this review will help to contribute to our knowledge of sediments in urban river basins and encourage further research so as to provide the necessary understanding for improved management.

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