Aquat. Sci. 64 (2002) 376–393
1015-1621/02/040376-18

Aquatic Sciences

Water Policy Article

Riverine quality at the Anthropocene: Propositions for global space and time analysis, illustrated by the Seine River

Michel Meybeck *

Université Paris VI, UMR Sisyphe, Place Jussieu, 75252 Paris cedex 05, France

Received: 23 June 2002; revised manuscript accepted: 25 September 2002

Abstract. The control of riverine quality (water, particulates) by human-related pressures is now a major feature of the Anthropocene era. A set of general typologies and approaches to address the complex relationships between pressures, environmental impacts and some of the related social responses is proposed here on the basis of various examples, among others the Seine basin. Riverine quality management is described through a dozen major types illustrated by river fluxes and riverine quality trends (⩾10 years). A successful restoration cycle, still seldom documented, is used as an example of the decomposition of the multiple social, societal and hydrological inertia and time lags, generally spanning several decades. Human impacts can also be described by finer temporal analysis, over hourly to year-on-year scales, and spatial analysis including classical longitudinal profiles, stream-order ranking and sediment pathways. The Seine river example illustrates the pressures from intensive agriculture, industrialisation, hydrological regulation and urbanisation with the impact of the World's second largest treated urban sewer discharge (from 8 million people in greater Paris).The impacts of the Paris megalopolis are much more widely spread than might be expected and include retro-impacts (in upstream reaches), distal (>100 km) and external impacts (outside of watershed). They are illustrated by specific spatial distributions of indicators of each particular phenomenon (organic pollution, metal contamination, xenobiotic occurrence, nitrate pollution, eutrophication). Although not comprehensive (acidification and salinization are not addressed here), such typologies should facilitate the comparisons between basins and phenomena at the regional and global scales.

Key words. River water quality; space-time analysis; global typology; Seine basin; Anthropocene.

Introduction

The anthropogenic forcing of the Earth Surface System is now equivalent or even greater than the natural one. The term Anthropocene has recently been used to qualify this fundamental change in the Earth's System, particularly regarding the climate (Crutzen and Stoermer, 2000). Although these authors identify Watt's invention of the steam engine (1784) as its starting point, another key date – 1950 – has been suggested (Meybeck, 2001a and c) for its full development, i.e. the point at which most indicators of human impacts (land use, dam construction, urban development, CO_2 increase, fertiliser use, nuclear energy impacts) show a sudden increase on most continents and/or have reached a global extension (Mackenzie and Mackenzie, 1995). This date is also the reference for ^{14}C dating (the so called Before Present ages).

Continental aquatic systems (CAS), here including soil water, streams and rivers, groundwater, lakes and estuaries, have been continuously modified since the early developments of agriculture, through land-use change, irrigation and navigation, the carrying of urban, mining and industrial wastes. These CAS modifications, including the creation of new types of aquatic systems such as canals, irrigation ditches, reservoirs, the alteration of natural hydrological regimes and the general degradation

* Corresponding author e-mail: Meybeck@ccr.jussieu.fr.
Published on Web: December 18, 2002

of CAS water quality, are now documented on all continents (Meybeck et al., 1989; Meybeck and Helmer, 1989; Degens et al., 1990; WHO/UNEP, 1992; Meade, 1995; Kimstach et al., 1998; Dynesius and Nilsson, 1994; Van der Weijden and Middleburg, 1989; Turner and Rabalais, 1991; Hu et al., 1998; Caraco, 1995; Seitzinger and Kroeze, 1998; Salomons et al., 1999; Peters and Meybeck, 2000; Von Bodungen and Turner, 2001; Neal and Whitehead, 2002).

Earth System scientists e.g., geochemists, physical geographers, oceanographers, hydrologists, consider the CAS in terms of natural storage, sources, processing and sinks of water and of all water-borne materials such as dissolved salts, organic and inorganic carbon, nitrogen and phosphorous, silica, metals, clay particles and sand. This river-borne material, continuously generated and transferred from land to oceans, plays multiple roles in the global regulation of the Earth at various times scales, from decades to hundreds of million years (Wollast et al., 1993; Mackenzie and Mackenzie, 1995; Ver et al., 1999; Berner and Berner, 1987). The alteration of these fluxes regulate the coastal-zone sediment balance, the ocean biogeochemistry and even the climate, through the continuous transfer of carbon from atmosphere to oceans.

In the field of social and economic sciences, the CAS, and particularly rivers, have been considered from different angles, ranging from religious and cultural values to economic resources characterized by supply/-demand ratio, water-quality criteria, international treaties or market prices. Most of these studies have pointed out the global-scale degradation of these water resources (Falkenmark and Lundqvist, 1995; Lundqvist, 1998). Recently, economists have opened up a new field with the inventory of the environmental services provided by aquatic systems and estimates of their values at the global (Costanza et al., 1997) and the local scale, e.g., coastal environments (Turner et al., 2001). These are major attempts to bridge the gap between the two scientific communities. Moreover, the long-term degradation of the water quantity and quality of CAS is now being considered in global-scale economic scenarios (Rotmans and de Vries, 1997).

This article is an attempt to establish further links between the two perspectives. The D.P.S.I.R. approach is used here: human activities are here termed "pressures", the "drivers" are: economic development, population increase, technical changes which result in modifications of river-water quality or change of "state" and, in turn, have impacts on societies which produce related "responses". This approach has already been used to study coastal-zone management by Turner et al. (1997) and Salomons et al. (1999). Here, these complex interactions are considered from various angles: (i) typology of trends in river material fluxes and concentrations, either dissolved or particulate, (ii) typology of river-quality management and successive time lags in management practices, (iii) decomposition of water quality restoration cycle, illustrating the social and societal responses to environmental impacts, (iv) typology of temporal variability of riverine quality both in natural and human-impacted conditions. These typologies result from the author's experience in comparing water-quality issues at a global scale (Meybeck et al., 1989; WHO/UNEP, 1991; Meybeck and Ragu, 1997; Meybeck, 2001a and b). (v) spatial and temporal analysis of some major river-quality issues (eutrophication, organic pollution, metal contamination, xenobiotic occurrence, nitrate pollution) as illustrated by the Seine River Basin. This analysis originates from a multidisciplinary study focused on this river since 1989, the Piren-Seine (Billen et al., 1994, 2001; Chevreuil et al., 1996; Garnier et al., 1998; Horowitz et al., 1998; Penven et al., 1998; Meybeck, 1998a; Teil et al., 1998; Barles, 2002; Barles et al., 2002; Thevenot et al., 2002), which has been presented in two books (Meybeck et al., 1998a; Garnier and Mouchel, 1999), and numerous reports.

Although they are not meant to provide management strategies, these typology attempts are destined to provide working tools for the temporal and spatial analysis of water quality and for comparisons of river basins at the regional to global scales combining Earth System and socio-economic analysis.

Typology of trends in riverine fluxes

Riverine flux trends can now be established for many rivers over 20 to 30 years and, for a few of them, over more than 50 years: most changes can be attributed to human pressures (Meybeck, 2001a). Riverine material fluxes are the product of concentration and water discharges. In the great majority of documented cases, they are linked primarily to changes in concentrations and can be schematised by a few trends, here normalised to the beginning of impacts (time T1), (Fig. 1), (Meybeck 2001a updated).

Flux trends are contradictory: many fluxes increase due to rising concentrations (types B, D, F, H, I, J, K, Fig. 1) but some of them actually decrease (types C, E, G) owing to a decrease of water discharge or to the biogeochemical and physical retention in an impoundment. Three types of flux decrease can be defined: (i) the hydrological changes caused by water diversion or use, mostly for irrigation, result in a *gradual decrease* of all water-borne fluxes (type G, Fig. 1). This is the case for many impounded basins in arid and semi-arid zones, e.g., the Colorado and Rio Grande, the Nile and Orange, Ebro and many mediterranean rivers, Amu Darya and Syr Darya, Indus, Huang He and Murray. Two other trend types are also linked to impoundment and do not concern

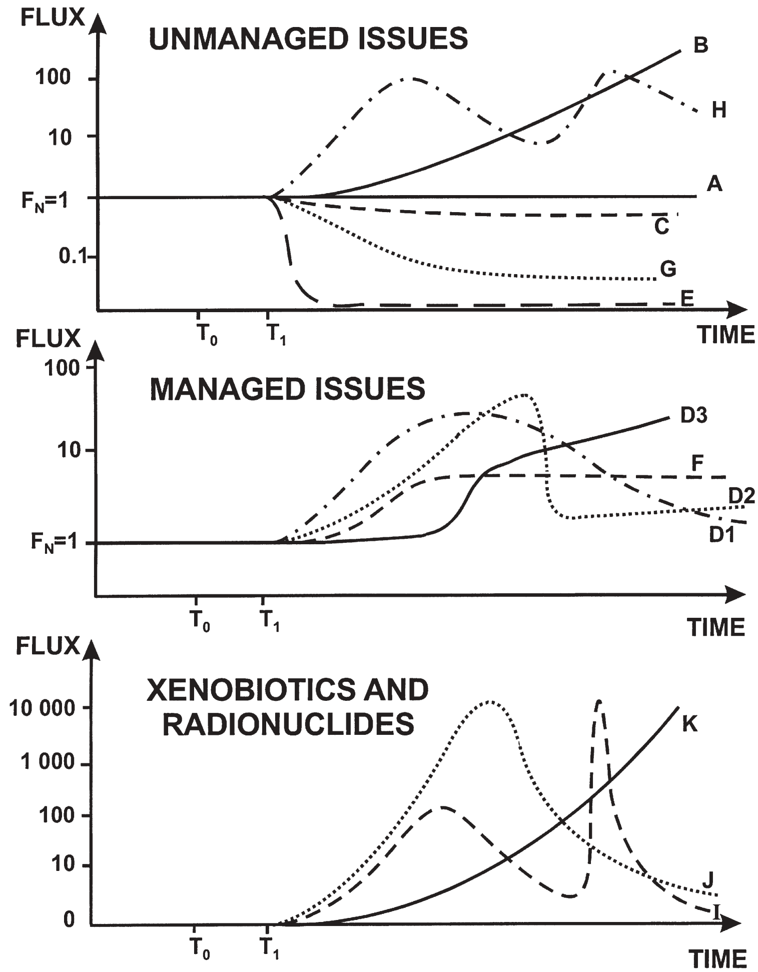

Figure 1. Types of river flux trends normalized to pristine fluxes (F_N) since the beginning of impacts (T_1) related to human pressures (T_0).
- Unmanaged issues: A: stable evolution, B: gradual increase, C: partial retention, E: complete retention, G: gradual decrease, H: multiple cycles (e.g. BOD5)
- Managed issues: D_1: bell-shaped control, D_2: stepwise improvement, D_3: stepwise degradation, F: stabilized contamination.
- Xenoproducts: I: multiple cycles of some radionucleids, J: total ban (as for DDT), K: gradual xenobiotic contamination.

the water use: (ii) *complete retention* caused by the settling of all particulate matter in reservoirs (type E), which exceeds 90% when the water residence time exceeds two months and might be responsible for trapping at least 30% of river particulates (Vörösmarty et al., 1997, 2002) and (iii) the *partial retention* resulting from the uptake of nutrients such as dissolved silica, phosphorus and nitrogen species (type C) in reservoirs. Silica uptake can also occur within a highly eutrophied river course, for example, the Rhine, Seine and Loire in Western Europe. As opposed to N and P compounds, this dissolved SiO_2 depletion is not compensated by anthropogenic sources. In these eutrophied and/or impounded rivers the Si/N ratio may decrease markedly and cause severe degradation of coastal-zone foodwebs, as is the case off the Mississippi and Danube deltas (Turner and Rabalais, 1991; Justic et al., 1995; Humborg et al., 1997).

Other trends of riverine fluxes and concentrations results from the continuous release of material through anthropogenic point and/or diffuse sources.

Very few elements are barely affected by human activities and present a *stable evolution* (type A): Ca^{2+}, Mg^{2+}, HCO_3^- in the Seine, and particulate Al, Fe, Si, Co, V, as observed in the Seine (Horowitz et al., 1998; Meybeck et al., 1998b). The *gradual increase* (type B) of many water-quality indicators corresponds to the development of these pressures, for example, Na^+, Cl^-, K^+, NO_3^-, SO_4^{2-} in the Seine (Meybeck et al., 1998b).

Bell-shaped control (type D1) characterises a successful and gradual control as that for heavy metals, NH_4^+ and PO_4^{3-} in the Rhine (Van der Weijden and Middleburg, 1989) or NO_3^- and PO_4^{3-} in Lake Geneva (Rapin et al., 1995). In some other rivers, the contamination has just started to decline in the 1990's, as in the case of metals in the Seine (Grosbois et al., 2003).

The stepwise improvement (D_2) is characteristic of a sudden decrease of contaminant concentrations (dissolved and/or particulate) in river systems. These trends are essentially caused by a drastic reduction in contaminant point sources such as that caused by major urban or industrial sewage treatments. In Northern France, the Espierre river, heavily impacted by sewage from a fertiliser plant, showed a marked drop in phosphate levels from 400 to 40 mg PO_4^{3-}/L in few months between 1983 and 1984 (Carpentier et al., 1994) as a result of pollution control. Such abrupt improvements can also be linked to the reduction or closure of economic activities during economic crises, viz., former socialist countries in Europe, e.g., the Elbe (Vink et al., 1999) or the Danube (A. Cosiasu pers. com.).

The stepwise degradation (D_3) is the symmetric evolution corresponding to the installation of major industries, collection of urban sewage without subsequent treatment or to sudden change in land use. This type is often found in long-term records of water quality, as in those of the Seine, monitored at Ivry since 1886 (Billen et al., 2001), and a very careful analysis is sometimes needed to explain these sudden increases. Land-use changes such as deforestation and conversion from grass to cropland may also lead to stepwise changes in river particulate fluxes or in nitrate fluxes. In both cases the step can last from a few weeks to one or two years.

Some trends are more complex: *multiple cycles* (type H) of contamination/improvement, often observed in very long series, such as those recorded since 1850 of the oxygen demand in the Thames estuary (Schwartz et al., 1990). Sediment archives of metal contamination provide numerous examples of multiple cycles, e.g., Neolithic gold mining in the Rio Tinto (Spain) (Leblanc et al., 1999) or the industrial impacts on the Rhine delta sediments, which peaked in 1930 and again in 1960 after a marked decrease during World War II (Middlekoop, 1997). Changes in land use and/or in agricultural practices can also show multiple cycles, provided that long-term records are available. The nitrate evolution in the Tweed river (UK) from 1956 to 1996 registered a decline in the late 80s when fertiliser application changed from autumn to spring (Neal, 2001).

Stabilised contamination (type F), i.e. very limited change over decades, may result from long-lasting impacts as those related to past mining activities or from long-term water-quality regulations as for example, the Colorado river salinity at the US-Mexican border or the Rhine chloride, both resulting from international treaties.

The xenobiotic pollutants – not produced naturally – have specific trends. The DDT evolution in the Northern Hemisphere CAS presents a gradual increase from flux zero, a marked peak then a decrease after its ban in the early 70s but can still be detected in trace amount in some rivers due to its great environmental persistence (*total ban*, type J). The herbicide contents, such as that of atrazine, increase generally (*gradual xenobiotic contamination*, type K) until they are finally regulated or the product is banned. For these less persistent products the decline may be rapid (a few years). Artificial radionucleids are often characterised by multiple cycles as for the artificial radiocesium in the Northern Hemisphere (*radioactive contamination*, type I) which peaked in 1962–1963, following trends of inputs into the atmosphere from nuclear tests, then again in 1986 after the Chernobyl accident.

Water-quality management strategies illustrated by trends in riverine quality indicators

The riverine flux trends can also be used to describe concentration trends in rivers, lakes and reservoirs, inputs to estuaries, and, apart from trend types E and G, to describe groundwater contamination. They result from a combination of human pressures, hydrosystem responses to human pressures, the development of social and societal awareness including advances in environmental science and in analytical techniques, the political decision processus, financial, technical or policy means of controlling these pressures or to carry out direct environmental remediation and finally, of the hydrosystem response to environmental control. This complex interaction will be discussed further. Both levels and trends of water-quality indicators can be used to characterise management strategies (Fig. 2, Meybeck 2001b updated).

In order to simplify the approach, only four levels of environmental quality are considered here: (i) <u>natural levels</u> (pristine or subpristine) characterised by natural indicator levels (C_N), (ii) <u>negligible impacts</u> on aquatic systems, observed through slight changes in water-quality indicators between C_N and a recommended level (C_R) at which all water demands are satisfied at minimum economic cost and at which no impact on human health or on the aquatic biota can be detected, (iii) <u>moderate impact</u> between (C_R) and (C_L), the limit at which severe economic losses implied by the water treatment and/or serious human health damage and/or serious and irreversible ecological damage occurs, (iv) <u>severe impact</u> above the C_L threshold.

Intermediate impacts can be found in some management rules (up to 10 classes of ecological damages have been proposed) but most operational classifications use 4 or 5 types. For example, the severe-impact level can be split into two types: the ultimate class then corresponds to a nearly abiotic system with very limited uses such as rivers converted into open-air sewers.

Several types of unmanaged and managed water quality situations can be described and illustrated by trends of water-quality indicators (Fig. 2). Several new

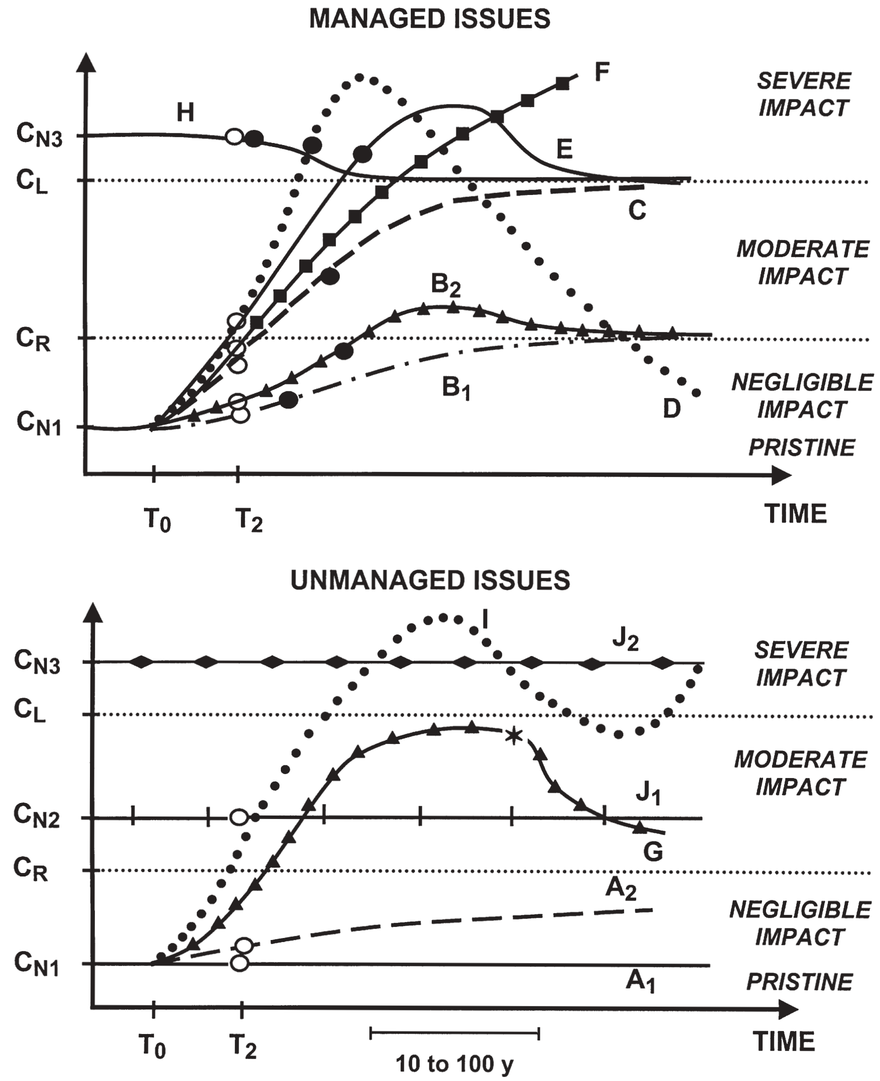

Figure 2. Typologies of river basin management strategies for water quality illustrated by trends in water quality. C_{N1}, C_{N2}, C_{N3}, natural, (C_R) recommended and (C_L) limit concentrations. T_0 = start of environmental pressures (×), T_2 = environmental impact detection (○), T_4 = start of environmental measures (●), unplanned decrease of environmental pressures (∗). A_1 and A_2: unecessary management, B_1: precaution management, B_2: delayed precaution management, C: maximum impact management, D: total ban, E: delayed pollution regulation, F: laissez-faire, G: unplanned improvement, H: natural pressure remediation, I: unperceived issue, J_1 and J_2: natural pressure endurance and natural pressure suffering.

types have been added to a preliminary version (Meybeck, 2001 b) in particular, to take into account the naturally occurring water-quality problems due to geologic or climate conditions (types J and H) when natural levels (C_{N2}, C_{N3}) exceed the water-quality criteria for specific uses or human health safety. Furthermore, prior to regular environmental monitoring, in the 60s and 70s in industrialised countries, in the 80s and 90s in developing countries, most problems went undetected and therefore, unmanaged.

Unmanaged issues

– *Unecessary management* (case A): the water quality is not affected by human pressures (eg Ca^{2+} or Mg^{2+} ions in many cases) (case A1), environmental or economic impacts are minimal and the rates of change are slow and predictable as for K^+ and, most of the time, Cl^- and SO_4^{2-} (case A2). This type is universal and related to major ions, some common and non toxic metals, and for refractory organic carbon derived from soil erosion.
– *Unplanned improvement* (case G) unexpected and/or unplanned decrease of contamination linked to the reduction (* on Fig. 2) of human pressures: closure of mines and industries, changes in technologies, economic crises. Such sudden improvements are commonly found for toxic compounds, metals and even nitrates in Transition Economy Countries e.g., in the

Elbe basin after 1989 (Vink et al., 1999). In some European rivers located in mining districts, a very slow metal decline (decades to centuries) is also noted in floodplain sedimentary archives resulting from the decline of mining activity since Roman times or the Middle Ages.

- *Unperceived issue* (case I): the deterioration of the water quality and/or its link to human pressure is not detected or perceived. Water-quality improvements are unplanned and the result of the balance between pressure and natural river basin response (e.g.: evolution of pollutant inputs, dilution, degradation of organic matter, sediment blanketing by uncontaminated particulates). Before the development of analytical chemistry in the early 1800s, followed by the very first water and sediment analyses and the establishment of the toxicity of some metals, this type of human-river relations was universal (Barles, 2002; Barles et al., 2002). The scientific and technical progress of analytical chemistry has always been the major regulator of the detection of water-quality problems: metal analysis by atomic absorption in the 1960s, organic carbon analysers in the 80s, gas chromatography for persistent organic pollutants and pesticides in the 80s, multielement analysis by ICP-MS in the 90's, etc.
- *Natural pressure endurance* (J_1) *and suffering* (J_2): in some rarely-found geological and climatic conditions the natural water quality is mediocre (C_{N2}), or unsuitable for most uses (C_{N3}), including drinking-water supply without major treatment (high salinity and total suspended solids; high natural concentrations in fluoride, arsenic, etc.). Depending on the availability of alternative water resources, the uses of such resources may lead to limited impacts (case J1) or to severe health and/or economic impacts (case J2). The last case is common in many arid and semi-arid countries and at sites with particular geological backgrounds (Bangladesh, Tanzania, China, Senegal, North Chile). It was generally not detected before 1970 for lack of appropriate water analyses.

Managed issues

- *Precaution management* (case B_1): environmental and economic impacts are kept to the minimum acceptable level (C_R). If action is taken too late, the level may first exceed this management target (B_2); such cases are found in highly developed and environmentally aware countries: Canada, Scandinavia or Switzerland, for instance, for eutrophication and phosphorus control. The Saint Lawrence basin management is a good example of such an approach at a regional scale.
- *Maximum impact management* (case C): targeted at the maximum acceptable limit (C_L), commonly chosen in international treaties (Colorado salinity at the US-Mexican border; Rhine chloride downstream of the Alsace Potash Mines outfall).
- *Total ban* (case D): ban on manufacturing and/or use of products; usually targeting xenobiotics only after severe problems have been detected, demonstrated and recognised by all stake holders (e.g., DDT ban in Europe and North-America, restriction on PCB use, recent regulation on atrazine in Western Europe, etc.); such regulations may be established at local (river basin), regional (e.g. Baltic Sea watershed) or global levels (e.g. CFC ban).
- *Delayed pollution regulation* (case E): established after a period of lack of management and subsequent severe impacts; targeted levels are usually the maximum acceptable ones for economic reasons. Very common until the mid-1900s for developed countries. Peak impacts were observed from 1960 to 1980 depending on issues and river basins (e.g., most Western European rivers including the Seine river).
- *Laissez-faire* (case F): although its severity is well established and even studied, the situation has not yet been adequately tackled for multiple reasons: lack of environmental awareness or commensurate societal consensus for the level of severity, shortage of financial means, lack of environmental regulations or of political will to enforce them. This type of situation is still common in some industrialised countries (e.g., around major mining districts in the USA, Russia and, probably, in many others) and in many fast developing countries although not often documented. In the Lot, a tributary of the Garonne river in France, at least 70 t of cadmium are stored in river bed sediments as a result of 100 years of mining activity only recently stopped (Blanc et al., 1999). Another example is the Idrija mercury mine in Slovenia, one of the biggest in the world, now closed, which was opened in 1492. Over hundreds of years of laissez-faire it has generated an enormous amount of Hg-rich sediments which gradually contaminate the Gulf of Trieste (Hines et al., 2001).
- *Natural pressure remediation* (case H): direct treatment of unsuitable natural water resources (desalinisation, defluorisation, removal of arsenic). The deionization of sea water, for drinking or agriculture, albeit exceptional, and the dilution by diversion of better-quality water (occurring between the Rocky Mountains States and some Prairie states in the USA) can also be put in this category. Generally the target is set up at the highest acceptable level (C_L).
- *Remediation of ancient contamination* (not shown): when riverine quality results from very ancient (100 to 1000 y) and long-lasting activities, the ambient geochemical background (C_A) of metals, nitrate and other elements, can exceed the severe-impact thresh-

old (C_L). In most cases there is no present-day economic or administrative entity directly linked to the contamination ("orphan pollution") which occurred at times of "unperceived issues". The water or particulates must be treated at high costs, as in the case of natural-pressure remediation. Ancient contaminations are commonly described in ancient mining districts as for Western Europe (Wales, Humber catchment, Rio Tinto, Idrija basin in Slovenia). They also occur in rural-community wells in India or Africa (nitrate contamination). This situation follows the Unperceived issues (I) and Laissez Faire (F) types. Corresponding restorations measures are rarely set up since they require environmental knowledge, societal concensus and financial means. A noted exception is for the US Superfund Sites which were highly contaminated.

– *Cyclic management* (not shown): over the very long term (50 to 100 y) the documented water-quality trend presents multiple cycles of deterioration and improvements resulting from the complex interactions of human pressures, environmental impacts and human responses.

The best example of cyclic management comes from the Thames estuary where two severe periods of near anoxia (1840–1880 and 1920–1965) correspond to phases of marked imbalance between urban pressures and societal responses in the management of urban wastes (Schwartz et al., 1990). Periods of satisfactory estuarine conditions (1880–1920 and 1970…) are the results of major environmental control measures (sewage collection and treatment, regulations, etc.) over several decades.

Unplanned improvements may also be cyclic as seen in the heavy-metal contamination trend observed since 2500BC in the Rio Tinto estuary (Leblanc et al., 2000) which shows multiple contamination peaks due to the mining activities over 4500 years, a typical example of ancient contamination.

The remediation of ancient severe impacts (cases D, E, I) is difficult to manage when the environmental degradation has lasted for decades or centuries: for CAS with long water residence times in some aquifers and lakes, and for all issues linked to particulate matter contaminations (polluted lake sediments and soils, mine tailings, former wastes dumps): remediation measures are generally very costly and sometimes not technically feasible.

Societal responses to aquatic-system degradation: example of a restoration cycle

The stages of societal and socio-economic responses to the degradation of CAS are complex and each management type can be decomposed into a specific set of stages. The full restoration and stabilisation cycle (types B_2, C, D, E, Fig. 2) presents a good example of societal responses, depending on pressure intensity, time constants and varying societal conditions (Meybeck, 2000b). The start of human pressure is set at time T_0. Then, the following stages can be distinguished (Fig. 3A):

1. *hydrosystem reaction to contamination* (T_0–T_1), depending on system size and contaminant pathways (e.g. dissolved vs particulate transfer); water and particulates residence time in hydrosystems is a key parameter;
2. *impact detection* (T_1–T_2) of hydrosystem changes by water users, scientists, specific citizen groups ("sentinels");
3. development of *societal awareness* (T_2–T_3): time for the development of general knowledge and understanding of the issue, sometimes delayed by lobbying from various social or economic groups;
4. *policy lag* (T_3–T_4): time for authorities or politicians to decide the appropriate action; such decision can be reached through environmental awareness of all stake holders (bottom-up consensus) or obtained and imposed by political decision (top-down);
5. *financial and technical lags* (T_4–T_5): time to fully enforce the decisions;
6. *hydrosystem reaction to restoration and remediation measures* to limit (T_5–T_7) or decrease the environmental and societal impacts (T_{6A}–T_{6B}).

Depending on the timing of impact detection, impact duration and remediation effectiveness, various threshold levels can be reached. If the environmental control is sufficiently effective, a limited level of maximum degradation (I_M, characterized by C_M and T_M) is reached, followed by an improvement phase. C_M can exceed the critical level C_L or can be kept below it if early and adequate controls are set up. When partial control strategies (Strategies C and E, Fig. 2) are practised, the new environmental state stabilises near the critical level C_L. If control is less effective, a critical period (T_{L1}–T_{L2}) can be defined ($C > C_L$) during which severe ecological impacts, human health impacts or economic restrictions are observed. If the recommended level (C_R) is again reached, the impairment period (T_{R1}–T_{R2}) characterises the total duration of impacts. In some aquatic systems of reduced size direct remediation can be undertaken (T_{6A}–T_{6B}) to accelerate the improvement of the conditions that would result from simple water and/or particulates renewal after the decline of contamination sources.

In reality, the full recovery cycle, i.e. a degradation phase (T_1–T_M) followed by a full recovery phase (T_M–T_{R2}), is rarely attained (Fig. 3B), since the common management types are: Maximum impact management, Delayed regulation, Laissez-faire (C, E and F). A successful cycle can be delayed or even stopped at all

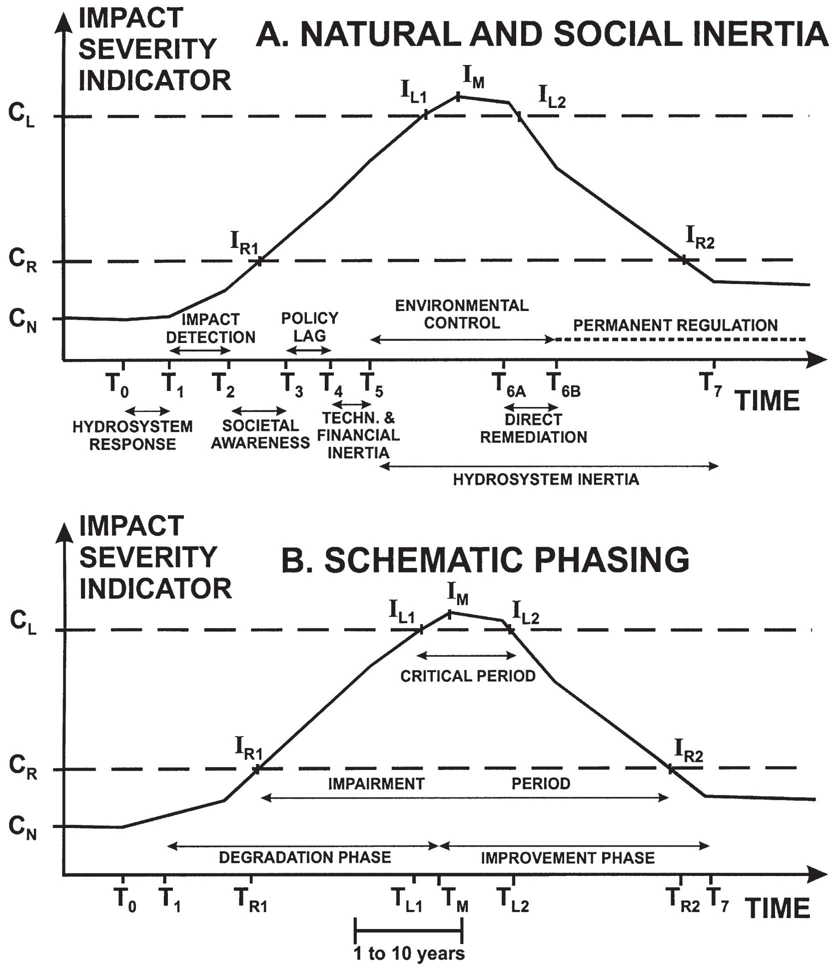

Figure 3. Successful restoration of water quality illustrated by a bell-shape trend in water quality. C_N, C_R, C_L: natural, recommended and limit concentrations or other environmental indicator (Meybeck, 2000 B modified). T_0: start of environmental pressure, T_1: first change of water quality, T_2: detection of change, T_3: established societal concern on issue, T_4: political decision concerning the issue, T_5: start of implementation of environmental measures, T_M: time of maximum impact, $T_{6A}-T_{6B}$: direct remediation measures, T_7: new steady state. Time scales vary according to issues and basin sizes.

stages: (i) the environmental state may not be known for lack of scientific knowledge (most issues prior to 1850–1900), because of lack of scientific consensus on the issue and/or its remedies (W. Salomons, pers. com.), lack of technical facilities (most new xenobiotics) (it takes about 10 y between the marketing of a new xenobiotic molecule used in industries or in agrochemicals and its regular monitoring at low cost and low detection limits, as for atrazine), lack of political will (e.g. Chernobyl fallout survey results were underestimated in France), (ii) impact detection may only be made when irreversible health effects have been observed, as for the mercury pollution in the Tokuyama Bay in Japan (Nakashini et al., 1989); (iii) the societal awareness and consensus may be counterbalanced by powerful lobbying as in the case of the eutrophication-phosphorus link which has been questioned in some European countries for one or two decades by the phosphorus industries; (iv) the policy lag can be enormous in order to protect one given socio-economic group, e.g., the agricultural nitrogen sources in Brittany (pigs and poultry farming) or in the Seine basin (intensive use of fertilisers) or chloride pollution in the Rhine from the Alsace Potash mines that daily discharge more than 10 000 t of NaCl; (v) the financial and technical lags can extend over several decades during the construction of sewage networks and treatment plants, particularly for megacities; (vi) the hydrosystem recovery is slower for longer water residence times (10 to 100 y and more for some aquifers and lakes), and for soil or sediment contamination.

In many examples of successful CAS restoration it has been necessary to act directly on the aquatic system (direct remediation $T_{6A}-T_{6B}$) by dredging the contaminated sediments (Tokuyama Bay), by inactivation of sediments below a layer of new sediments, by direct chemical treatment of water as in Lago di Orta, Italy (Bonacina and Baudo, 2001), siphoning anoxic waters etc. Such measures accelerate the recovery process but are rarely applied to large water bodies. Examples of time scales in the restoration of Tokuyama Bay (Nakashini et al., 1989) and Lake Geneva (Rapin et al., 1995) are compared in Meybeck (2001b and c): the impairment period (T_{R1}–T_{R2}) is only 25 y for Tokuyama Bay where polluted sediments could be removed but will be over 50 y in Lake Geneva (Rapin et al., 1995) despite an exceptionally fast political response (T_2–$T_5 \approx 10$ y) although the critical level of nutrients has never been reached. For Lake Orta the degradation phase (total destruction of fauna) was very short, 3 years, and the improvement phase now exceeds 70 years despite a direct liming of the lake 50 y (T_1–T_6) after the impact (Bonacina and Baudo, 2001).

Each environmental history of CAS can be, in principle, decomposed in a similar way. Yet the order of societal responses may greatly differ from those presented here (e.g. lack of sentinels, delayed political decisions, lack of restoration techniques etc). The application of the precaution management may even result in political decision (T_4) before any adverse impact is detected (T_1), thus benefiting from environmental experiences from other basins. The decomposition proposed here should now be tested in other river basins to assess the management strategies for each water quality issue in relation to their social and societal contexts.

Typology of temporal water quality variations and their alteration by human impacts

In addition to trends, generally defined on annual averages over at least a decade, other water quality variations are also observed at much shorter time scales ranging from sub hourly to annual. They are very informative as to natural processes occurring in catchments (water pathways, tributary mixing, weathering types, biogeochemical cycles linked to terrestrial and aquatic biota) and human-influenced impacts (diffuse and point sources of pollution, water discharge regulations, land-use changes, etc.). These temporal variations are schematised in Figure 4 from a spatial perspective since some of them are highly dependent on catchment size while others are not.

This temporal-spatial link is particularly strong in the variations linked to water discharge (Q), whether natural or anthropogenic. In very large river basins Q variations are very smooth, corresponding to a seasonal flood, while in rainfall-fed small river systems the annual hydrograph is very irregular with multiple Q peaks corresponding to each rainstorm and rapid variations of water quality at the daily scale.

In natural river basins, most of the water-quality controls are linked to Q, i.e., to the relative proportions of groundwater, surface water and subsurface water and therefore depend on the basin size (control factor A, Fig. 4). In larger basins, particularly heterogeneous ones, the tributary mixing is an additional factor that depends on basin size (control B). Daily variations originate from snow or ice melting cycles (control C) and from nychthemeral variations (day/night) linked to the production/respiration cycles of phytoplankton and bacteria, particularly for the lower reaches and at lake outlets (control D). Seasonal cycles (control E), often masked by the previous variations, result from cycles of vegetation growth or of soil biogeochemistry. Year-on-year variations are linked to climate variability (control F) and variations over 100 to 1000 y depend on climate change. Event-based variations, not shown here, have been reported (snow melt flushing of soils, forest fires, landslides, volcanic eruption, etc.).

Anthropogenic forcing is added to the previous ones (Fig. 4B). From the shortest to the longest, the impacts can be listed as follows: urban rainstorm runoff (impact A), which typically last for a few hours, and accidental spills (impact B); daily cycles of urban or industrial waste production (impact C); daily cycles of regulated water discharge, generally associated with reservoirs (impact D); enhanced eutrophication in impounded or navigated river reaches or during summer low flows with marked nychthemeral O_2 and pH cycles as in the Loire basin (Moatar et al., 1999) (impact E); weekly or monthly cycles of waste production (impact F) of reservoir operation as in many Swiss rivers (OFEH, 1995) (impact G); seasonal production of wastes, seasonal cycles of atmospheric pollution, seasonal use of agrochemicals and seasonal water discharge regulations (impacts H); 10-year programmes for waste product collection and treatment (impact I), gradual land-use changes (impact J) and other trend types. Other event-based variations linked to human activities can be observed (pesticide flushing at the first rain after spreading, strikes, etc.). Many of these impacts have a magnitude that depends on the dilution capacities of the river system, i.e. on catchment size, e.g., impacts A, B, I and J. Weekly and monthly cycles, that are not observed in natural conditions can be termed *xenocycles*. A good example of such cycles can be observed for pH, O_2, temperature and conductivity at many water-quality observation stations in Switzerland (OFEH, 1995) downstream of the Alpine reservoirs.

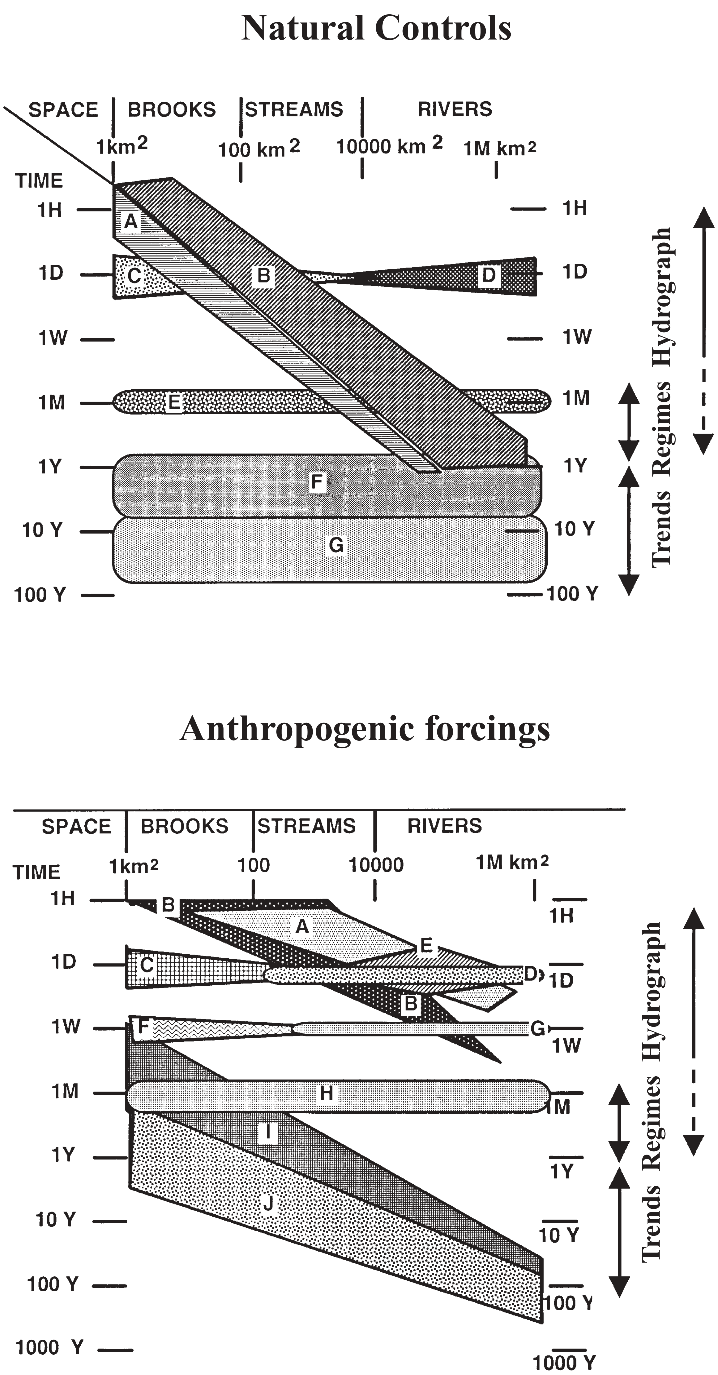

Figure 4. Time and space scales of riverine quality variations. Natural controls: A: water pathway through soils and groundwaters; B: tributaries mixing; C: daily freeze/thaw cycles; D: aquatic biota production/respiration cycles; E: seasonal soil and vegetation processes; F: interannual hydrometeorological variabilities; G: climate variability and climate change. Anthropogenic forcings: A: urban storm runoff; B: accidental spills; C: daily waste inputs cycles; D: daily water dicharge regulation; E: enhanced riverine eutrophication; F: weekly waste inputs cycles and water discharge regulation (G); H: seasonal production of wastes, of agrochemical uses, atmospheric pollution and water discharge regulation; I: reduction of pollution point sources; J: land use changes.

Spatial complexity of riverine responses to pressures: Megacity impacts

The megacity impact illustrated by Paris and the Seine River

Megacities constitute one of the most prominent features of environmental change in the second half of the 20th century. They provide some of the biggest point sources of aquatic pollution. Common sense usually restricts the impact of a major city to a 10–100 km downstream river reach (*proximal impact*). In reality retro-impacts can also be observed on upstream reaches and downstream impact can affect other downstream river reaches and estuaries (*distal impacts*) and the coastal zone (*external impacts*), far away from the megalopolis. Depending on human pressures and their environmental responses, the spatial distribution of megalopolis impacts can be very complex. The schematic example of the Seine river is used to illustrate this (Fig. 5 a).

The Seine basin (65 000 km^2 upstream of the estuarine limit, 17 million people, 480 m^3/s) is moulded by the enormous weight of the Paris megalopolis (2300 km^2 of urban + suburban area, 10 million people connected to sewage systems, 25 m^3/s of treated sewage

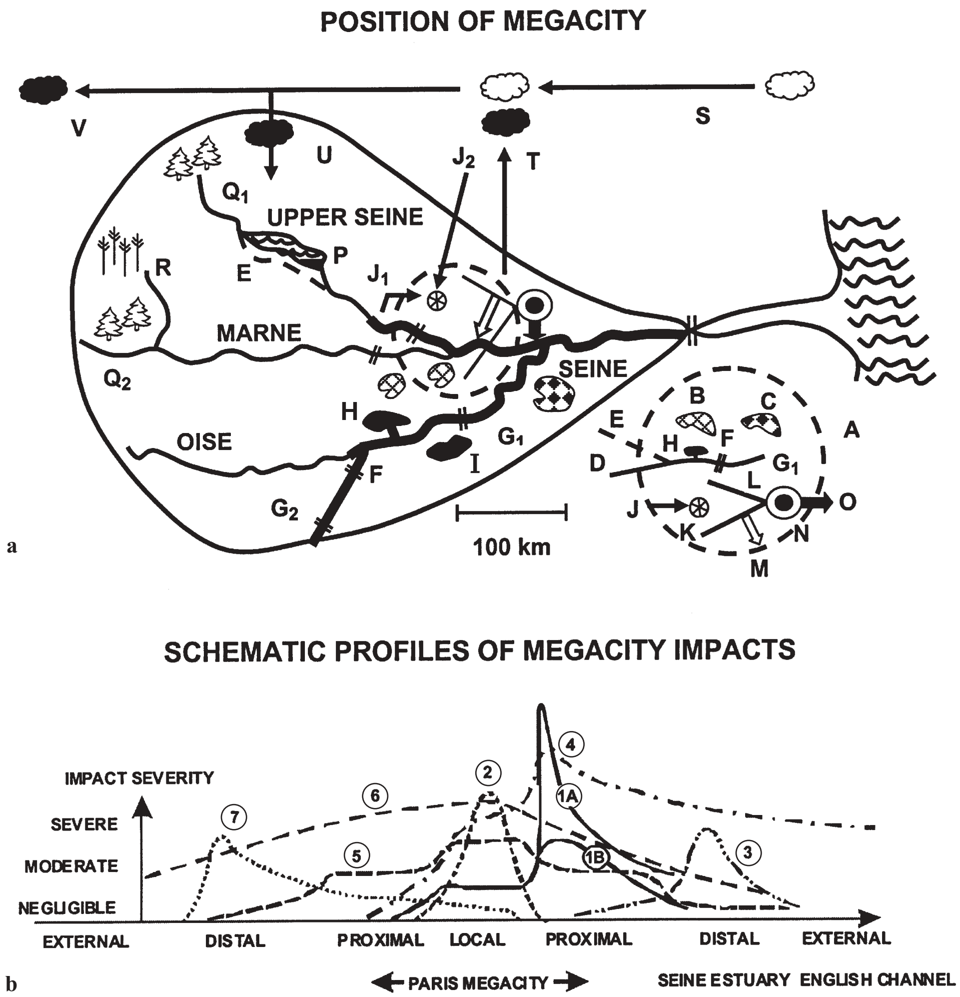

Figure 5. Schematic spatialization of megacity impacts: example of Paris and the Seine river basin. Positions of megacity: A Megacity limits; B land fills; C Area of urban sludge reuse in agriculture; D Natural river reaches; E reaches bypassed by major reservoirs (P); F navigation locks; G_1 channelized reaches and canals (G_2); H sand pits opened on the rivers and isolated (I); J surface waters diverted to drinking water plant (K) sometimes originating from outside of the river basin (J_2); L and M, separated and combined sewer systems; (O) treated sewage from the Seine-Aval plant (N); P reservoirs and artificial river reaches bypassing natural course; Q_1 and Q_2 remaining forested headwaters, used to qualify natural background; R agricultural headwaters; S atmospheric inputs to the basin; T local atmospheric emissions and deposition (U); V atmospheric outputs from the basin. Impacts profiles: 1 faecal pollution and oxygen demand during dry weather (1A, 1950–1970; 1B, present day); 2 combined sewer overflow impact during rainy days; 3 delayed impact of nitrification in the estuary and bacterial degradation of riverine organic matter; 4 metal contamination (1980's); 5 present artificialisation of river course and hydrological regulation; 6 atmospheric fallout contamination; 7 past impact of timber rafting from Morvan Hills (Q_1). Other secundary urban impacts in the basin or estuary have not been represented.

released by the Seine-Aval treatment plant, the world's second largest one with 8 million equivalent people) (Meybeck et al., 1998; Meybeck 1998). The megalopolis sewage system is combined in its oldest parts, designed in the 19th century, and separate in recent networks. Most of the collected and treated sewage is injected into the Seine, 75 km downstream of central Paris, a few km upstream of its confluence with the Oise where the river is of stream-order 8. The lower parts of the Seine river and its major tributaries, the Oise and the Marne, are channelized, dredged and locked for navigation purposes (Fig. 5a, F and G) and within Paris, the river banks are isolated from the natural alluvial plain by concrete or stone walls.

In order to protect Paris from major floods and droughts, a series of large reservoirs totalling 725 million m^3 and 92 km^2 were built on the upper tributaries (Seine, Marne, Aube) between 1991 and 1996, 200 to 300 km upstream of Paris (Garnier et al., 2001). The summer low flows in Paris, originally as low as 40 m^3/s, are now sustained so as to reach 90 m^3/s. The three major reservoirs were constructed outside the river thalweg, thus bypassing the natural river course, where a minimum flow is maintained (Fig. 5a, E and P). Such regulation and artificialisation of the river course is not new: from the 1600s to 1920 the transport of wood for fuel from the Morvan forest, some 400 km upstream of Paris, has been continuous and had a maximum impact on the upper reaches (Barles et al., 2002).

Profiles of the severity of Paris impacts can be scaled from various physical, chemical and biological indicators and compared (Fig. 5b, profiles 1 to 7) according to the four-grade scale previously presented (Figs. 2 and 3). The profiles of other major impacts related to diffuse agricultural pollution sources or to the impact of smaller cities (the second city in the catchment is only 400000 equivalent people) are omitted in this graphic and discussed in the following section. Impact types include the physical deterioration of aquatic habitat, the changes in natural hydrological regimes, the inputs of pollutants from dry-weather and wet-weather sewers, the leaching of agrochemicals, the atmospheric pollution inputs and the delocalized pollution.

An example of past impact of human pressures is schematized on profile 7 for timber floating, an important activity between the Morvan forested headwaters and Paris from 1600 to 1920 (Barles et al., 2002). More recent physical impacts of Paris on the continental aquatic system include hundreds of sand pits, excavated in the alluvial Holocene deposits, upstream and downstream of Paris, some of them in direct communication with the river course where they can serve as fish refuges, in case of water-quality deterioration, and as spawning grounds. The sand and gravel demand is attributed to the development of the megalopolis and its commercial activity (profile 5, Fig. 5b): at present, 50 million tons of alluvial material are now extracted annually, i.e. about 70 times the present suspended load exported to the Seine estuary. The artificialization of the riverine habitat is maximum within the Parisian reach.

The impacts of hydrological regulation and reservoir management are greatest in the bypassed reaches and downstream of reservoirs on the Marne, Aube and Seine (Garnier et al., 1998). All these impacts have been aggregated (profile 5, Fig. 5b).

Despite the sustained river low flows and a treatment efficiency of collected organic pollution of up to 90%, a major water-quality impact is still observed in summer downstream of the Seine-Aval treatment plant (profile 1B) (Servais et al., 1998). From the 1950s to early 1970s, before the full secondary treatment of the Paris wastewater was put in place, this massive point source of pollution in the river resulted in extensive hypoxic events during the summer, over a 200 km reach from the Seine-Aval plant to the upper estuary (profile 1A).

During summer rainstorms, the direct overflow of combined sewers within the Paris reach produces sudden peaks in bacterial counts, ammonia and oxygen demand (profile 2, Fig. 4) (Servais et al., 1998). Their impacts on the river have recently been reduced by direct injection of oxygen where appropriate.

In addition to these local and proximal impacts, a delayed impact of treated Paris wastewater has been noted in the estuary, some 200 km downstream of the treatment plant (profile 3), qualified here as a distal impact. It is linked to the slow estuarine nitrification of urban-derived ammonia released by the Seine-Aval plant (Billen et al., 2001) and decomposition of algal detritus resulting from lower-basin eutrophication, which starts at stream-order 4 and peaks at orders 6 to 8 (Billen et al., 1994). In the lower reaches, the total pigments may exceed 150 μg/L or 4.5 mg/L of highly degradable algal particulate organic carbon (POC).

The longitudinal profile of heavy metals and of some persistent organic pollutants in suspended matter is strongly correlated to the population density in smaller stream-orders upstream of Paris (this impact is not represented in Fig. 5b and is discussed in the next section). The impact of the Paris megalopolis (profile 4, Fig. 5b) on heavy metals and POPs (persistent organic pollutants) is well marked. It shows an increase from suburban reaches upstream of Paris with a peak downstream of the Seine-Aval plant. Heavy metals and POPs are then slightly diluted by the less contaminated particulates from the Oise tributary (Grosbois et al., 2003; Meybeck et al., 2003). As most of these products are poorly reactive, they are scarcely modified in the estuarine zone (apart from cadmium which is partially solubilized) and transferred directly to the English Channel (external impact).

In addition to these impacts, along the river course and flood plain, from the headwaters to the estuary and coastal zone, the atmospheric contamination related to the emissions of the Paris megalopolis (T, Fig. 5 b) is well marked, although probably decreasing (Thévenot et al., 2002) and affects the whole basin (profile 6, Fig. 5 b), particularly downwind of the prevailing West-East atmospheric circulation, i.e., in upstream river reaches (U, Fig. 5 b). This contamination can be regarded as a distal impact of Paris and the long-range atmospheric transfer of Paris pollutants to other river basins can be regarded as an external impact.

Finally, another category of Paris impacts on aquatic systems concern the *delocalized impacts* related to (i) recycling of wastes outside the basin (most of the present lead flux in the Seine basin is due to car batteries, which are now recycled outside the basin, with all possible environmental impacts) (Thévenot et al., 2002), (ii) exports of pollutant-containing manufactured products and, more importantly, (iii) imports of basic products from other river basins. The footprints of Parisians on the environment can therefore be found across the world: affecting the production of pulp and paper from taiga woods, furniture from tropical forests, metal ores from many countries, phosphorus ore from Morocco, tropical foodstuffs, etc.

Stream-order ranking and water quality

Anthropogenic impacts on riverine water quality are often illustrated by longitudinal profiles from the headwaters to the river mouth such as those for the Mississippi (Meade et al., 1995) or the Rhine (Van der Weijden and Middleburg, 1987). Such approach was used in the previous section to describe the multiple impacts of a megalopolis on a receiving river using the example of Paris and the Seine river main channel (Fig. 5 A and B). Many other impacts related to diffuse sources of atmospheric pollutants (not covered in this article) and agricultural pollution from dispersed sources such as small villages, industries, mining operations should be taken into account in addition to large city impacts. A corresponding degradation of the water quality is often observed in very small catchments or headwaters (Strahler stream-orders 1 to 3 at the 1/100 000 scale).

In the stream order approach, systemically tested on the Seine river, all equivalent stream-orders are pooled, from very small streams (order 1) to the river mouth (order 8 in the Seine) according to the Strahler definition (Billen et al., 1994; Meybeck et al., 1998). Their median concentration, normalized to the natural background – defined on forested headwaters – are schematized in figure 6 for natural products – major ions, metals, nutrients, organic carbon. For xenobiotics e.g., atrazine, the most widely used herbicide, and PCBs, natural background values cannot be defined and the average level at river mouth (C_{OUT}) is used for normalization.

Regarding most chemical indicators, their concentrations increase regularly from stream-orders 1 or 2, exposed to agricultural impacts mostly in areas with a human population density of < 10 p · km^2 (AGR in Fig. 6) to orders 6 and 7 within the Paris megalopolis (density increasing from 100 to 250 p · km^2) on the middle Seine course, the lower Marne and the lowest part of the Oise (see Fig. 5 a). The final reach (order 8) downstream of the Seine-Oise confluence remains at constant pressure.

The dilution capacity of pollution point sources is very limited in this river system, characterised by medium runoff (annual average 8 l · s^{-1} · km^{-2}, 1 l · s^{-1} · km^{-2} for summer low flows) and very low sediment yields (10 t · km^{-2} · y^{-1}). Therefore, the Seine basin is sensitive to impacts of small towns and cities, generally located on stream-orders 3 to 6 (suburban Paris excluded), where the population density increases from 20 to 100 p · km^{-2}. Most water-quality indicators presented in the upper figure 6 show a significant increase from the natural background to rural streamorders (1 to 3) characterized by a population density of 20 p/km^2 (Na^+, SO_4^{2-}; Cu, Hg, Pb and Zn; NH_4^+ and PO_4^{3-}). These indicators then increase regularly with the population density with regard to background levels. The maximum impact is measured downstream of the Paris sewage treatment plant, some 60 km downstream of central Paris (see Fig. 5 a and 5 b).

Several indicators of major water-quality criteria do not follow this schematic profile (middle Fig. 6). The nitrate maximum, around 5 mgN/L, is observed in agricultural basins of stream-orders 1 and 2, exposed to heavy fertiliser use, then it becomes nearly stable and independent of the population density. Algal particulate organic carbon (POC measured by total pigments) increases in orders 4 to 5, where the phytoplankton growth is not limited by inorganic N (up to 7 mg N/L) or P (0.5 mg P/L) but by the dilution of algal cells (Billen et al., 1994, Garnier et al., 1998). As a result of eutrophication, an increase in algal POC and uptake of dissolved silica by diatoms are observed. The fertilisers impact is observed for K^+, particulate phosphorus (PP) and cadmium, which is commonly found in P-fertilizers. However most of the PP and Cd fluxes exported to the estuary stem from urban and remaining industrial sources. The impact of atmospheric contamination is difficult to assess for lack of adequate monitoring, yet it is likely for Cl^- and possible for SO_4^{2-}. Metal contamination of atmospheric fallout is well assessed but the related impact on river water and particulates (e.g. in forested headwaters) is not obvious.

The profile of atrazine, a common herbicide, is determined here from the maximum values, not yearly averages as for previous indicators (lower Fig. 6). Maximum levels are generally observed after the first heavy spring rain following spreading. In stream-orders 1 and 2, the

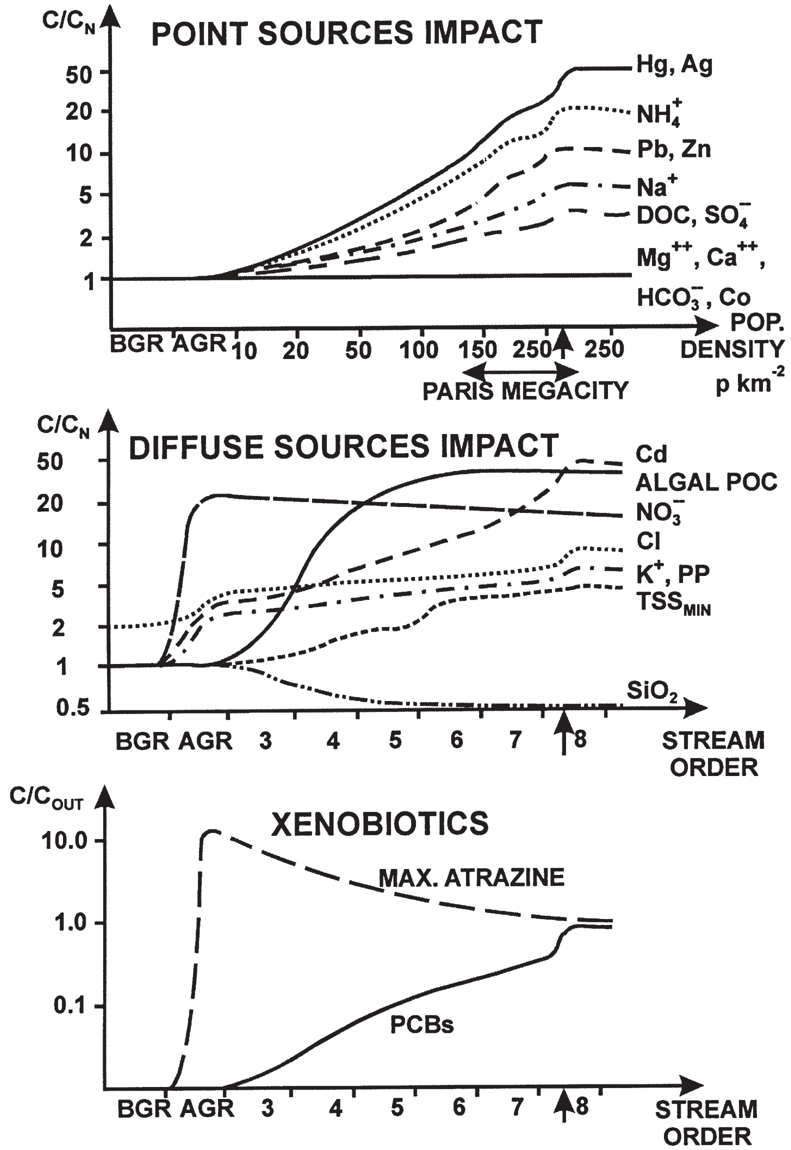

Figure 6. Schematic ranking of median water quality indicators in the Seine River by stream orders from headwaters to river mouth (500 km). BGR = natural background (forest streams orders 1 to 3); AGR = stream orders 1 and 2 impacted by agriculture.
– Upper: Natural products with marked impacts of point sources and/or urbanization: annual median concentration in water and annual mean contents in particulates, normalized to natural background (C_N) generally increasing with population density
– Middle: Natural products with marked impacts of diffuse sources (atmospheric fallout for Cl^-; agriculture for NO_3^-, K^+, and particulate phosphorus = PP) and of eutrophication occuring on orders 4 to 8 (algal POC production and dissolved silica uptake); increase of minimum TSS due to eutrophication, navigation, and sewer inputs. Annual means or median normalized to natural background (C_N).
– Lower: Xenobiotics. Mean PCB's and maximum spring atrazine normalized to river mouth mean value (C_{OUT}).
Note log-scales. Arrow indicates the release of Paris treated sewers.

maximum values may reach 10 mg/m^3, far exceeding the drinking-water standard of the European Union (1 mg/m^3 for the sum of pesticides) and decrease to about 1 mg/m^3 in orders 5 and 6, where the Paris city water intakes, are located (Fig. 5a). As rainwater leaching is not synchronous in the whole Seine catchment, the maximum atrazine level at the river mouth is at least one order of magnitude lower than those observed on small orders. The profile of PCBs, a family of xenobiotic products used as indicators of past industrial and urban pressures, reflects its origins with peaks downstream of the Paris megalopolis (lower Fig. 6).

As previously mentioned, some water-quality indicators are barely impacted by human activities. In the Seine basin, this concerns Ca^{2+} and HCO_3^-, which are generally regulated by calcite saturation, Mg^{2+}, a few particulate trace metals such as Be, Co, V and most major particulate metals (Al, Fe, Ti, Na, K) where the amounts

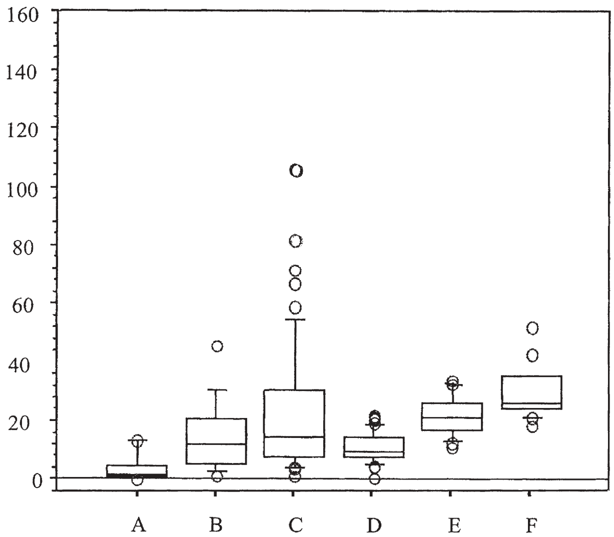

Figure 7. Statistical distribution of a water quality indicator using a stream order ranking. Example of a polymetallic pollution index (MPI based on Cd, Cu, Hg, Pb and Zn) in the Seine catchment (Meybeck et al., 2003). A Forested catchments only (stream orders 1 to 3); B Non forested catchments, (orders 2 to 3); C Orders 4 and 5; D orders 6 and lower Oise (order 7); (E) order 7 (Seine within Paris); (F) order 8 (downstream of Paris). Contamination scale: 2 to 5, very low; 5–10, low; 10–20, medium; 20–50, high; 50–100, very high.

generated by human activities are slight compared to those from naturally occurring sources.

These profiles are averaged over the period 1985–1995. Yet the Seine basin decontamination is progressing rapidly. For instance the Cd levels in sewage particulates received by the Seine-Aval treatment plant have decreased by a factor of 10 in the last 20 years. Most heavy metals show a similar trend, attributed to major abatements in industrial sources, particularly plating. The mercury trend, also decreasing, is more complex (Meybeck et al., 2003).

Stream-order rankings of water-quality indicators, presented in figure 6, are based on medians or average values for each order. In reality, the spatial distribution can be more complex for some issues related to point sources. This is illustrated by figure 7 where the statistical distribution of an integrated index of metallic pollution established for the Seine (MPI, based on Cd, Cu, Hg, Pb and Zn) is reported for six clusters of stream-orders. The MPI distributions based on 163 samples of river particulates, mostly fresh flood deposits, are presented as box-plots ($MPI_{10,25,50,75}$ and MPI_{90} for each cluster (Meybeck et al., 2003)). The lower quantiles (MPI_{10} and MPI_{25}) regularly increase from forested catchments (cluster A), where the contamination is negligible (MPI < 1), to values exceeding 25 for order 8, i.e., in the lower Seine downstream of Paris (cluster F). The other quantiles ($MPI_{50,75,90\%}$) show maximum values exceeding 50 for clusters C, i.e., in medium stream-orders (4 and 5). The maximum upper decile (MPI_{90}) which exceeds 30, a value characteristic of a high contamination, is also observed for cluster F (downstream of Paris).

The very high MPI values observed in orders 4 and 5 can be attributed to suburban catchments where the population density ranges from 500 to 1000 $p \cdot km^{-2}$. Such contamination level is extremely high: it corresponds to an average increase factor of more than 10 for the 5 metals with regard to the theoretical geochemical background as determined by their mineralogical assemblage. However it must be noted that background metal levels in the Seine catchment are exceptionnally low due to the abundance of sedimentary carbonates and quartz sands deposits, particularly for Cd and Hg (Grosbois et al., 2003). Therefore the MPI should be clearly differenciated from an indicator of toxic effects on the biota.

Conclusion

Human impacts on river basins, particularly regarding water- and particulate-matter quality, are expressed in multiple ways depending on physical geography factors, on basin development history, sometimes extending over thousands of years, on current human pressures and on societal responses to environmental protection.

A schematic comparison of river-quality trends (variations over at least a decade) permits a characterisation of some major types of river-quality evolution which can be linked to a dozen key societal responses in terms of river quality management.

The analysis of temporal variations of river quality at a finer scale – hourly to interannual also reveals the multiple impacts of human activities that distort natural variations and add new ones, for example, the xenocycles that do not exist in natural conditions. These variations (daily, nychthemeral cycle, flood cycle, seasonal cycle, etc.) provide valuable information on sources, pathways and levels of contamination, which characterise each phenomenon. Global-scale typologies of contamination-water discharge relationships and of daily flux patterns such as those established by Williams (1989) for Total Suspended Solids (TSS) concentrations and by Meybeck et al. (2002) for TSS loads, should also be established for inter-basin comparisons.

A spatial analysis of water-quality issues was attempted for the Seine Basin, on the basis of a set of innovative studies and surveys; it illustrates the diversity and complexity of each intervening factor. Compared to most rivers, the Seine Basin is very homogeneous in terms of hydroclimate, relief, soils and land use but it harbours one of the world's greatest point sources of pollution (250 000 people for 1 m^3/s at natural low flow downstream of the Paris megalopolis). The Seine basin suffers a combination of most human pressures, except mining, resulting in very severe quality problems as to heavy metals, organic pollution, xenobiotics – pesticides and persistent organic pollutants – eutrophication and nitrate pollution, compared to other world rivers (Meybeck et al., 1989, Meybeck et al., 1998 a). Their spatial distribution was studied through schematic mapping, longitudinal profiles along the main river course, stream-order analysis and particulate matter analysis. Each river-quality characteristic has a different space distribution where, in most cases, areas of minimum impacts can be found and eventually used as references in orienting basin restoration. A detailed analysis of the Paris megalopolis impact also shows unexpected retro-impacts on upstream reaches and far-reaching, or distal, impacts on the estuary and the coast, thus considerably increasing the conventional proximal impacts, within and immediately downstream of the city. In future studies, delocalised and external impacts, i.e. outside the catchment, will have to be addressed.

The typologies proposed here and the various approaches to spatial analysis of river-basin quality are not comprehensive, e.g., the basin acidification by atmospheric inputs and salinization are not treated here, but they cover a wide range of issues as organic pollution, eutrophication, nitrate pollution, metal contamination, xenobiotic occurrence, acidification, bacterial contamination (Chapman, 1992), which can be decomposed into complex interactions between man and river.

Each one of these issues can be characterized by specific temporal and spatial distributions such as those described schematically here. They result from complex relationships between human pressures, impacts on river quality, impacts on basin or coastal-zone economy, human and aquatic biota health and the related societal responses (Turner et al., 2001; von Bodungen et al., 2001). The corresponding river quality management strategies can be described into at least a dozen of types.

For each issue successful cycles of river-quality restoration should be decomposed into a set of inertia and time lags, biogeochemical and hydrological, scientific, social, political, technical and regulatory, in order to describe the evolution of the river quality. In most documented case studies, the basic temporal unit for riverine evolution is a decade.

Our typologies and hypotheses should now be compared with the corpus of well-documented river basins. The Seine example used here is probably exceptional from the point of view of the range of studied issues and available long-term records (the earliest regular surveys date back to 1886). Other river basins have been studied in great detail and/or monitored over a long time, and even modelled as, for example, the Mississippi (Meade et al., 1995), the Saint Lawrence (Centre Saint Laurent, 1996), the former Soviet Union rivers (Kimstach et al., 1998), the Po (Water Research Institute, 1991), the Humber estuary basin (Neal et al., 2000), the Rhine and the Elbe (Behrendt and Opitz, 2000; de Wit, 2000; Vink et al., 1999; Van der Weijden and Middleburg, 1989), the Murray (MDBC, 1999), the Mgeni in South Africa (Kienzle et al., 1997) as well as many others.

Such synthetic analyses are meant to facilitate the comparison of river basins at the regional and global scales for a twofold purpose: (i) determine the position of fluvial systems in the Earth System as promoted by the emerging Joint Water Group of the WCRP, IGBP, IHDP and DIVERSITAS programmes, by the SCOPE Silica and SCOPE Nitrogen programmes, by the new IOC-river nutrients programmes, (ii) determine the evolution of fluvial systems at the regional scale and its societal responses, as promoted by the European Union and many Regional Sea programmes of the UNEP, and as carried out for some years by the IGBP-LOICZ (EUROCAT, SAM CAT and others) and by START, and, more recently, by the Unesco-HELP programme.

Acknowledgments

Although this paper results from a personnal analysis, the author is particularly indebted to his colleagues from the Piren-Seine programme led by G. de Marsily and G. Billen for multiple discussions on the Seine Basin. River typologies have been established by continuous efforts within the Sequana project (CNRS, France), the IGBP-BAHC core project and the EUROCAT European programme through exciting discussions with Sabine Barles, Laurence Lestel, Charles Vörösmarty, Wim Salomons, Art Horowitz, Colin Neal, A. Cosiasu and many others.

References

Barles, S., P. Benoit and L. Lestel (eds.), 2002. Analyse rétrospective du système Seine. Synthèse 1998–2001. Rapports Piren Seine 2002, UMR CNRS 7619 Sisyphe, Univ. Paris 6, 57 pp.

Barles, S., 2002. L'invention des eaux usées: l'assainissement de Paris 1780–1930. In: C. Bernhardt, G. Massard-Guilband (eds.). The Modern Demon. Pollution in Urban and Industrial European Societies, Presses de l'Univ. Blaise Pascal, Clermont-Ferrand, pp 129–156.

Behrendt, H. and D. Opitz, 2000. Retention in river systems: dependence of specific runoff and hydraulic load. Hydrobiologia **410:** 111–122.

Berner, R. A. and E. K. Berner, 1987. The Global Water Cycle, Prentice Hall, 455 pp.

Billen, G., J. Garnier and Ph. Hanset, 1994. Modelling phytoplankton development in whole drainage networks: the River-Strahler model applied to the Seine river system. Hydrobiologia **289:** 119–137.

Billen, G., J. Garnier, A. Ficht and C. Cun, 2001. Modeling the response of water quality in the Seine River Estuary to human activity in its watershed over the last 50 years. Estuaries **24:** 977–993

Blanc, G., Y. Lapaquellerie, N. Maillet and P. Anschutz, 1999. A cadmium budget for the Lot-Garonne System (France). Hydrobiol **410:** 331–341.

Bonacina, C. and R. Baudo (eds.), 2001. Lake Orta: a case study. J. Limnology Special Issue **60:** 213–300.

Caraco, N. F., 1995. Influence of human population on P transfers to aquatic systems: a regional case study using large rivers. In: H. Thiessen (ed.). Phosphorus in the Global Environment, SCOPE series, John Wiley and Sons, pp 235–244.

Carpentier, P., A. Leprêtre and J. Prygiel, 1994. Evolution et prévision des concentrations en phosphates dans les eaux superficielles en Artois-Picardie. In: Rencontres Hydrologiques Franco-Roumaines, Unesco Techn. Doc. Hydrol. Paris.

Centre Saint Laurent, 1996. Rapport-Synthèse sur l'état du Saint Laurent. Vol. I. L'écosystème du Saint Laurent, Environnement Canada et Editions Multimondes, Montréal, 600 pp.

Chapman, D. (ed.). Assessment of the quality of the aquatic environment through water, sediment and biota. Chapman and Hall, London, 550 pp.

Chevreuil, M., M. Blanchard, M. J. Teil, A. M. Carru, P. Testard and A. Chesterikoff, 1996. Evaluation of the pollution by organopolychlorinated compounds and metals in the water and in the zebra mussels (*D. polymorpha Pallas*) of the Seine River. Water, Air, Soil Pollution **88:** 371–381.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton and M. van den Belt, 1997. The value of the world's ecosystem services and natural capital. Nature **387:** 253–260.

Crutzen, P. J. and E. F. Stoermer, 2000. The "anthropocene". IGBP Newsletter **41:** 17–18.

Degens, E. T., S. Kempe and J. E. Richey, 1990. Biogeochemistry of Major World Rivers. SCOPE 42, John Wiley, N. Y., 356 pp.

De Wit, M., 2000. Modelling nutrient fluxes from sources to river load: a macroscopic analysis applied to the Rhine and Elbe basins. Hydrobiologia **410**: 123–130.

Dynesius, M. and C. Nilsson, 1994. Fragmentation and flow regulation of river systems in the northern third of the world. Science **266:** 753–762.

Falkenmark, M. and J. Lundqvist, 1995. Looming water crisis: new approaches are inevitable. In: L. Ohlsson (ed.), Hydropolitics: Conflicts Over Water as a Development Constraint, Zed Books, London, pp 178–212.

Garnier, J., G. Billen, Ph. Hanset., P. Testard and M. Coste, 1998. Développement algal et eutrophisation dans le réseau hydrographique de la Seine. In: M. Meybeck, G. de Marsily, E. Fustec (eds.). La Seine en son bassin. Elsevier, Paris, pp 593–626.

Garnier, J. and J. M. Mouchel (eds.), 1999. Man and River Systems. The Functioning of River Systems at the Basin Scale. Hydrobiologia, special volume **410:** 355 pp.

Garnier, J., B. Leporcq, N. Sanchez and X. Philippon, 1999. Biogeochemical mass-balances (C, N, P, Si) in three large reservoirs of the Seine Basin (France). Biogeochemistry **47:** 119–146.

Grosbois, C., M. Meybeck and A. Horowitz, 2003. Trace and major elements geochemistry of the Seine River particulates: I, Spatio-temporal distribution of Cd, Cu, Hg, Pb and Zn (1981–2000) (submitted to Sci. Tot. Env.).

Hines, M. E., M. Horvat and J. Faganelli (eds.), 2001. Mercury in the Idrija Region and the Northern Adriatic, Portoroz workshop, RMZ-Materials and Geoenvironment (special issue), **48:** 1–259.

Horowitz, H., M. Meybeck., Z. Idlafkih and E. Biger, 1998. Variation of trace elements contamination in the Seine river basin using floodplain deposits and streambed sediments. Hydrological Processes **13:** 1329–1340.

Humborg, C., V. Ittekot, A. Cociasu and B. Von Bodungen, 1997. Effect of Danube river dam on Black Sea. Biochemistry and ecosystem structure. Nature **386:** 385–388.

Justic, D., N. N. Rabalais, R. E. Turner and Q. Dortch, 1995. Changes in nutrient structure of river-dominated coastal waters: stoichiometric nutrient balance and its consequences. Estuar. Coastal Shelf Science **40:** 339–356.

Kienzle, S. W., S. A. Lorentz and R. E. Schultze, 1997. Hydrology and Water quality of the Mgeni Catchment. Water Research Commission, Pretoria, South Africa, WRC Rept. TT 87/97, 88 pp.

Kimstach, V., M. Meybeck and E. Baroudy (eds.), 1998. A Water Quality Assessment of the Former Soviet Union. E&FN Spon, London, 611 pp.

Leblanc, M., J. A. Morales, J. Borrego and F. Elbaz-Poulichet, 1999. A 4500 years old mining pollution in Spain. Economic Geology **95:** 655–662.

Lundqvist, J., 1998. Avert looming hydrocide. Ambio **27:** 428–433.

Mackenzie, F. T. and J. A. Mackenzie, 1995. Our Changing Planet, An Introduction to Earth System Science and Global Environmental Change. Prentice Hall, 387 pp.

MDBMC, 1999. The Salinity Audit of the Murray-Darling: A 100-year Perspective. Murray-Darling Basin Ministerial Council. Canberra, ACT, pp 39.

Meade, R. H. (ed.), 1995. Contaminants in the Mississippi River 1987–1992. U. S. Geol. Survey Circular 1133, Reston, Va., 140 pp.

Meybeck, M., 1998a. Man and river interface: multiple impacts on water and particulate chemistry illustrated in the Seine River Basin. Hydrobiologia **373/37:** 120.

Meybeck, M., 1998b. Surface water quality: global assessment and perspectives. In: H. Zebibi (ed.), Water, a loaming crisis, Unesco Techn. Doc. Hydrology **18:** 173–187.

Meybeck, M., 2001a. River basin under anthropocene conditions. In B. von Bodungen, K.Turner (eds.), 2001. Science and Integrated Basin Management. Dahlem workshop series, Wiley, pp 275–294.

Meybeck, M., 2001b. Global alteration of riverine geochemistry under human pressure. In: E. Ehlers and T. Krafft (eds.). Understanding the Earth System: Compartments, Processes and interactions, Springer, pp 97–113.

Meybeck, M., 2001c. Integration of earth systems and human dimensions in continental aquatic systems. IHDP Update:14–16.

Meybeck, M., D. Chapman and R. Helmer (eds.), 1989a. Global Fresh Water Quality: A First Assessment. Basil Blackwell, Oxford, 307 pp.

Meybeck, M. and R. Helmer, 1989. The quality of rivers: from pristine stage to global pollution, Global and Planetary Change **1:** 283–309.

Meybeck, M., G. Friedrich, R. Thomas and D. Chapman, 1992. River Monitoring. In: D. Chapman (ed.) Assessment of the Quality of Aquatic Environment Through Water, Biota, and Sediment, Chapman and Hall, London, pp 239–316.

Meybeck, M. and A. Ragu, 1997. Presenting GEMS-GLORI, a compendium of world river discharges to the oceans. Int. Ass. Hydrol. Sci. Publ. **243:** 3–14.

Meybeck, M., G. de Marsily and E. Fustec (eds.), 1998a. La Seine en son bassin. Elsevier, 750 pp.

Meybeck, M., J. M. Mouchel, Z. Idlafkih, V. Andreassian and S. Thibert, 1998b. Transferts d'eau, de matières dissoute et particulaire dans le réseau fluvial. In: M. Meybeck, G. de Marsily, E. Fustec (eds.), La Seine en son bassin, Elsevier, pp 345–389.

Meybeck, M., Z. Idlafkih, N. Fauchon and V. Andreassian, 1999. Spatial and temporal variability of Total Suspended Solids in the Seine basin. Hydrobiologia **410:** 295–306.

Meybeck, M., A. Horowitz and C. Grosbois, 2003. Trace and major elements geochemistry of the Seine River particulates: II, Distribution patterns of an integrated metallic contamination index (submitted to Sci. Tot. Env.).

Meybeck, M., L. Laroche, H. H. Dürr and J. P. M. Sgvitski, 2002. Global variability of daily Total Suspended Solids and their fluxes in rivers. Global Biogeochem. Cycles (in press).

Middelkoop, H., 1997. Embanked floodplains in the Nederland. Nederlandsee Geografische Studies, 224, Utrecht, 342 pp.

Moatar, F., A. Poirel and C. Obled, 1999. Analyse des séries temporelles de mesure de l'oxygène dissous et du pH sur la Loire au niveau du site nucléaire de Dampierre (Loiret). Hydroecologie Appl. **11:** 127–151.

Nakashini, H., M. Ukita, M. Sekine and S. Murakami, 1989. Mercury pollution in Tokuyama Bay. In: P. G. Sly and B. T. Hart (eds.) Sediment/Water Interaction. Kluwer, pp 198–208.

Neal, C., 2001. The water quality of eastern UK rivers: the study of a highly heterogeneous environment. In: D. A. Huntley, G. J. L. Leeks and D. E. Walling (eds.), Land-Ocean Interaction, IWA Publishing, pp 69–104.

Neal, C. and P. G. Whitehead (eds.), 2002. Water Quality Functioning of Lowland Permeable Catchments: Inferences from an Intensive Study of the River Kennet and Upper River Thames. Sci. Total Environment **282–283**, 512 pp.

Office Fédéral de l'Economie Hydraulique, 1995. Annuaire Hydrologique de la Suisse. OFEH, Bern.

Penven, M. J., T. Muxart, F. Bartoli, P. Bonté, D. Brunstein, C. Cosandey, V. Gouy, S. Irace, T. Leviandier and S. Sogon, 1998. Petits bassins ruraux et pollutions diffuses. In: M. Meybeck, G. De Marsily, E. Fustec (eds.). La Seine en son bassin. Elsevier, pp 159–210.

Peters, N. E. and M. Meybeck, 2000. The linkage between freshwater availability and water-quality degradation: cyclical and cascading effects of human activities. Water International **25:** 185–193.

Rabalais, N. N. and R. E. Turner, 2001. Hypoxia in the Northern Gulf of Mexico: description, causes and change. In: N. N. Rabalais and R. E. Turner (eds.): Coastal Hypoxia. Consequences for Living Resources and Ecosystems. Am. Geoph. Union, Costal and Estuarine Studies **58:** 1–36.

Rapin, F., P. Blanc, J. P. Pelletier, G. Balvay, D. Gerdeaux, C. Corvi, J. Perfetta and C. Lang, 1995. Impacts humains sur les systèmes lacustres: exemple du Léman. In: R. Pourriot, M. Meybeck (eds.), Limnologie Générale, Masson Paris, pp 806–840.

Rotmans, J. and B. de Vries, 1997. Perspectives on global change. The TARGETS approach. Cambridge Univ. Press, 462 pp.

Salomons, W., K. R. Turner, L. D. de Lacerda and S. Ramachandran (eds.), 1999. Perspectives on Integrated Coastal Zone Management. Springer, 386 pp.

Schwartz, H. E., J. Emel, W. J. Dickens, P. Rogers and J. Thompson, 1990. Water Quality and flows. In: B. L. Turner et al. (ed.). The Earth as Transformed by Human Action. Global and Regional Changes. In: The Biosphere Over the Past 300 years. Cambridge University Press, pp 253–270.

Seitzinger, S. P. and C. Kroeze, 1998. Global distribution of nitrous oxide production and N inputs in freshwater and coastal marine ecosystems. Global Biogeochem. Cycles **12:** 93–113.

Servais, P., G. Billen, J. Garnier, Z. Idlafkih, J. M. Mouchel, M. Seidl and M. Meybeck, 1998. Carbone organique: origines et biodégradabilité. In: La Seine en son bassin, Elsevier, Paris, pp 483–529.

Teil, M. J., M. Blanchard, M. Chesterikoff and M. Chevreuil, 1998. Transport mecanisms and fate of polychlorinated biphenyls in the Seine River (France). Sci. Total Environment **218:** 103–112.

Thevenot, D., M. Meybeck and L. Lestel, 2002. Métaux lourds, des bilans en mutation. Rapport de Synthèse 1998–2001, Piren Seine, UMR CNRS 7619 Sisyphe, University Paris VI, 78 pp.

Turner, B. L., W. C. Clark, R. W. Kates, J. F. Richards, J. T. Mathews and W. B. Meyer, 1990. The Earth as Transformed by Human Action, Cambridge University Press, 713 pp.

Turner, R. E. and N. N. Rabalais, 1991. Changes in Mississippi river water quality this century and implications for coastal food webs. BioScience **41:** 140–147.

Turner, R. K., C. Perrings and C. Folke, 1997. Ecological economics: Paradigm or perspective. In: J. van den Berg and J. van der Strutten (eds.). Economy and Ecosystems in change; Analytical and Historical Approaches. Edward Edgar, Aldershot, pp 42–55.

Turner, R. K., I. J. Bateman and W. N. Adger (eds.), 2001. Economics of Coastal and Water Resources: Valuing Environmental Functions. Kluwer, 342 pp.

Van der Weijden, C. H. and J. J. Middleburg, 1989. Hydrogeochemistry of the River Rhine: long term and seasonal variability, elemental budgets, base levels and pollution. Water Research **23:** 1247–1266.

Ver, L. M., F. T. Mackenzie and A. Lerman, 1999. Biogeocheminal responses of the carbon cycle to natural and human pertur bations: past, present and future. Am. J. Science **299:** 762–801.

Vink, R. J., H. Behrendt and W. Salomons, 1999. Point and diffuse source analysis of heavy metals in the Elbe drainage area: comparing heavy metal emissions with transported river loads. Hydrobiologia **410:** 307–314.

Von Bodungen, B. and R. K. Turner (eds.), 2001. Science and Integrated Basin Management. Dahlem Workshop Series, Wiley, 380 pp.

Vörösmarty, C., M. Meybeck, B. Fekete and K. Sharma, 1997. The potential impact of neo-castorization on sediment transport by the global network of rivers. Int. Ass. Hydrol. Sci. Publ. **245:** 261–282.

Vörösmarty, C. and M. Meybeck, 1999. Riverine transport and its alteration by human activities. IGBP Newsletters **39:** 24–29.

Vörösmarty, C. J., M. Meybeck, B. Fekete, K. Sharma, P. Green and J. Syvitski, 2002. Anthropogenic sediment retention: major global-scale impact from the population of registered impoundments. Global Biogeochem. Cycles (in press).

Walling, D. E., 1999. Linking land use, erosion and sediment yields in river basins. Hydrobiologia **410:** 223–240.

Water Research Institute, 1991. Water Quality in the Po Basin, Water Research Inst. Special Issue. Quad. Ist. Ric. Acque, 92, CNR, Roma, pp 300.

WHO/UNEP (M. Meybeck, R. Helmer, M. Dray, H. El Ghobary, A. Demayo, S. Ramadan, K. Khan, P. J. Peterson and J. Jackson), 1991. Water quality, progress in the implementation of the Mar del Plata Action Plan. WHO/UNEP (UN Conference on Water, Dublin, 1992), 79 pp.

Williams, G. P., 1989. Sediment concentrations versus suspended matter discharge during hydrologic events in rivers. J. Hydrol. **111:** 89–106.

Wollast, R., F. T. Mackenzie and L. Chou, 1993. Interactions of C, N, P and S. Biogeochemical Cycles and Global Change. Springer-Verlag Berlin, 521 pp.