Chapter 6* - Rivers

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6.1. Introduction

Rivers are the most important freshwater resource for man. Social, economic and political development has, in the past, been largely related to the availability and distribution of fresh waters contained in riverine systems. Major river water uses can be summarised as follows:

- sources of drinking water supply,
- irrigation of agricultural lands,
- industrial and municipal water supplies,
- industrial and municipal waste disposal,
- navigation,
- fishing, boating and body-contact recreation,
- aesthetic value.

A simple evaluation of surface waters available for regional, national or trans-boundary use can be based on the total river water discharge. The Colorado River, USA is an example where extraction of water has virtually depleted the final discharge to the ocean. The flow has been used almost completely by negotiated extraction and distribution to nearby states. Any increase in extraction and use would require diversion of a similar water quantity to guarantee the minimum flow required to meet all the water demands of the region.

Upstream use of water must only be undertaken in such a way that it does not affect water quantity, or water quality, for downstream users. Use of river water is, therefore, the subject of major political negotiations at all levels. Consequently, river water managers require high quality scientific information on the quantity and quality of the waters under their control. Provision of this information requires a network of river monitoring stations in order:

- to establish short- and long-term fluctuations in water quantity in relation to basin characteristics and climate,
• to determine the water quality criteria required to optimise and maintain water uses, and

• to determine seasonal, short- and long-term trends in water quantity and quality in relation to demographic changes, water use changes and management interventions for the purpose of water quality protection.

As with all freshwater systems, river quality data must be interpreted within the context of a basic understanding of the fluvial and river basin processes which control the underlying characteristics of the river system. Similarly, the design of the monitoring network, selection of sampling methods and variables to be measured must be based on an understanding of fluvial processes as well as the requirements for water use.

6.2. Hydrological characteristics

6.2.1 River classification

Rivers are complex systems of flowing waters draining specific land surfaces which are defined as river basins or watersheds. The characteristics of the river, or rivers, within the total basin system are related to a number of features. These features include the size, form and geological characteristics of the basin and the climatic conditions which determine the quantities of water to be drained by the river network.

Rivers can be classified according to the type of flow regime and magnitude of discharge (see below for further details). The flow regime may be subject to considerable modification by natural impoundments, lakes, dams, or water storage (see Chapter 8). Flow characteristics may also be changed by canalisation, or requirements for water uses, such as withdrawal for irrigation or other water supply needs, or by changes in flood characteristics due to modifications of the soil infiltration as a result of agriculture and urbanisation.

The classification of rivers according to their discharge is generally more satisfactory but has not, to date, been completely defined and accepted. However, there are certain specified discharge rates which are widely used to characterise river discharges and their annual variations. These include the average peak discharges, the monthly or annual average discharge and the average low discharge. A size classification based on discharge, drainage area and river width is given in Table 6.1. The distinctions are arbitrary and no indication of the annual variability in discharge is given. River discharge, particularly in arid and sub-tropical regions, may range from zero in the dry season to high discharge rates in large rivers during the rainy season. Very large rivers may also traverse many climatic zones and can have less variability than might be expected for the climatic conditions at the final point of discharge, such as the Mississippi and Nile rivers.

Rivers drain watersheds of varying dimensions. As indicated in Table 6.1, this area is directly related to the river discharge and width. Efficient drainage is achieved by means of a dendritic network of streams and rivers. As these increase in size from small to large, and then to the main river channel, the “order” in which they appear is a function of the
watershed size. An example of an “ordered” network and the relationship of stream orders to other river characteristics is given in Figure 6.1.

Figure 6.1  A. The relationship between stream orders and hydrological characteristics using a hypothetical example for a stream of order 8: (1) Watershed area (A); (2) Length of river stretch (L); (3) Number of tributaries (n); (4) Slope (m m$^{-1}$); B. Stream order distribution within a watershed

Table 6.1 Classification of rivers based on discharge characteristics and the drainage area and river width

<table>
<thead>
<tr>
<th>A River size</th>
<th>Average discharge ($m^3$ s$^{-1}$)</th>
<th>Drainage area ($km^2$)</th>
<th>River width (m)</th>
<th>Stream order$^1$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Very large rivers</td>
<td>&gt; 10,000</td>
<td>&gt; 10$^6$</td>
<td>&gt; 1,500</td>
<td>&gt; 10</td>
</tr>
<tr>
<td>Large rivers</td>
<td>1,000-10,000</td>
<td>100,000-10$^6$</td>
<td>800-1,500</td>
<td>7 to 11</td>
</tr>
<tr>
<td>Rivers</td>
<td>100-1,000</td>
<td>10,000-100,000</td>
<td>200-800</td>
<td>6 to 9</td>
</tr>
<tr>
<td>Small rivers</td>
<td>10-100</td>
<td>1,000-10,000</td>
<td>40-200</td>
<td>4 to 7</td>
</tr>
<tr>
<td>Streams</td>
<td>1-10</td>
<td>100-1,000</td>
<td>8-40</td>
<td>3 to 7</td>
</tr>
<tr>
<td>Small streams</td>
<td>0.1-1.0</td>
<td>10-100</td>
<td>1-8</td>
<td>2 to 6</td>
</tr>
<tr>
<td>Brooks</td>
<td>&lt; 0.1</td>
<td>&lt; 10</td>
<td>&lt; 1</td>
<td>1 to 3</td>
</tr>
</tbody>
</table>

$^1$ Depending on local conditions

River systems represent the dynamic flow of drainage water, which is the final product of surface run-off, infiltration to groundwater and groundwater discharge. The general relationships between these and the nomenclature for a river transect are summarised in Figure 6.2.
6.2.2 Velocity and discharge

Hydrological characteristics are determined by velocity and discharge. The velocity (sometimes referred to as flow) of the river water is the rate of water movement given as m s\(^{-1}\) or cm s\(^{-1}\). The discharge (m\(^3\) s\(^{-1}\)) is determined from the velocity multiplied by the cross-sectional area of a river. Cross-sectional area fluctuates with the change in water level or river stage. Similarly, a direct relationship exists between water level and velocity and measurement of level can be transformed directly into velocity. A velocity calibration can then be derived. With known cross-sectional area, the discharge \(Q\) can be derived by measurement of the water level. The level can be obtained using an appropriate gauge placed on the bank, or on another suitable structure in the river. It can often be measured automatically giving a continuous record (hydrograph) of the changing levels or direct discharge of the river. Further details of measurement of velocity, level and discharge are available in WMO (1980) and Herschy (1978). The discharge of a river is the single most important measurement that can be made because:

- it provides a direct measure of water quantity and hence the availability of water for specific uses,
- it allows for the calculation of loads of specific water quality variables,
- it characterises the origins of many water quality variables by the relationship between concentrations and discharge (see section 6.3.3), and
- it provides the basis for understanding river basin processes and is essential for interpreting and understanding water quality.
River regime

The size and geological formation of a watershed determines the river discharge regime. The discharge and its annual, as well as long-term, fluctuations are primarily influenced by the characteristics of the drainage basin. Climatic, meteorological, topographical and hydrological factors play a major role in the generation of river discharge.

Small watersheds usually result in low median discharges with extremely large ratios of peak and low discharge. In temperate humid climates, the annual variations between minimum and maximum discharges may reach two orders of magnitude. Larger watersheds produce more uniform discharges. Large rivers with a relatively uniform discharge regime during the year show a rather constant ratio of average peak and low discharge. Large seasonal variations in the discharge can be equalised and transformed into rather uniform discharges by the presence of reservoirs, storage dams or natural lakes along the river course. Examples include some of the alpine rivers which are impounded for hydroelectric power generation. The Rhine river, which is more than 1,000 km in length, has maintained a low-to-peak discharge ratio of 1:15 in the lower reaches, constantly over the last 20 years. This is because different parts of the catchment area contribute high discharge at different times of the year.

The principal factor causing large fluctuations in discharge is climate, which determines the distribution of rainfall over the year. The variability and resulting non-uniformity of discharge is moderate in temperate humid climates, but extreme for rivers in savannah areas and in certain sub-tropical regions. The composition and structure of the sub-soil are also important factors. Large differences can be observed between porous rocks, clays, marshy soils and fissured rocks. Such geological conditions of the drainage basin might cause variations in the discharge rates by a factor of two and in a few cases even more.

Vegetation also exerts an influence on the generation of river discharge because it largely determines the quantity of surface run-off. Fluctuations in discharge can be dampened by vegetation cover. In areas with little or no vegetation, rainfall results in immediate surface run-off.

For problems of water quality management, such as the disposal of wastewaters into rivers, low discharge conditions are used as the basis for the design of treatment facilities and control of the maximum permissible effluent disposal to rivers. At any higher river discharge rate, the ecological effects of a polluted effluent in a river are less harmful. Drought conditions can be critical for rivers serving as a water source for urban water supply.
Discharge characteristics

Most rivers are characterised by a condition called base flow or base discharge. This is the minimum amount of water moving within the individual river system, and in most cases is controlled by groundwater discharge. Figure 6.3 shows the characteristics of a storm controlled river basin by comparing daily discharge and daily rainfall in 1986 for the Venoge river draining the Jura mountains into Lake Geneva. Considerable increases in discharge $Q$ above the base level occur during storm events. The increase in discharge is not only a function of the storm intensity but is also related to the duration of rainfall, soil water saturation, etc.

Discharge characteristics of rivers are greatly modified by the nature of the watershed. Changes in the infiltration rate to the groundwater system change the water run-off characteristics of the watershed. Loss of forest cover and the proportion of exposed bedrock to deep soils and sediments, all have profound effects on discharge characteristics. These changes are shown in Figure 6.4 which illustrates two river basins of the same dimensions but different infiltration rates, following a storm event of the same duration and intensity. With high infiltration within the watershed (Figure 6.4A) the peak flow or flood stage has a much lower discharge than in the basin of low infiltration (Figure 6.4B). In addition, the hydrograph in Figure 6.4A shows an increased duration of the flood stage indicating that the run-off from the watershed extends over a longer time scale. When a watershed is changed from type A to type B (e.g. by urbanisation) the river channel characteristics, which were created under high infiltration rates, are no longer adequate to accommodate the high and rapid flood discharges. Under these conditions flooding becomes a major urban problem which may lead to overloading of the storm water collection facilities and sewage treatment systems. As a result, these discharges to rivers and coastal areas may contain higher than usual levels of organic
matter, faecal bacteria and toxic substances. The resultant increases in biochemical oxygen demand (BOD) (see section 3.5.3) can lead to fish kills, and the high concentrations of faecal bacteria may restrict water use, including bathing in coastal waters. The construction of storm water ponds or reservoirs allows the collection and slow release of storm waters to the main river channel or coastal waters (see Chapter 8). Further details are available in UNESCO (1988).

Figure 6.4 Theoretical hydrographs of a rain storm event of the same intensity in two basins of equal size but with different infiltration rates: A. High infiltration rate e.g. sandy soil, forest area; B. Low infiltration rate e.g. base crystalline rock, urbanised basin. The increase in base flow is different in each case

Discharge may also be controlled by factors other than storm conditions. In all cold latitudes and mountainous regions the effects of freezing and thawing in glaciers are particularly important. Figure 6.5 shows the mean and maximum monthly discharges of the Alpine Rhône river, Switzerland at various locations from upstream (Figure 6.5 A) to the river mouth at Lake Geneva (Figure 6.5D). Low discharge occurs during glacial freezing in the winter and high discharge occurs during the summer ice melt between May and October. The variations between winter low discharge and summer high discharge are less pronounced downstream. This is due to the increasing influence of storm related run-off and snow melt in the middle altitudes on the lower reaches of the river, and to less variation in discharge as a function of increasing basin area.
Figure 6.5 Long-term monthly means and maximum discharges at four stations on the Alpine Rhône river, Switzerland which is influenced by glacial melt (Modified from Burrus et al., 1990)

An additional feature which can be inferred from the hydrographs of the Alpine Rhône is the relationship of discharge to basin area. The larger the basin, the greater the damping effect on the variability in the discharge. This dependence is illustrated in Figure 6.6 by three theoretical hydrographs for river basins of different dimensions but with the same water regime.

6.2.3 Fluctuations in suspended solids

The concentration of total suspended solids (TSS) in rivers increases as a function of flow. Particles are derived by sheet, bank and gully erosion in the watershed and by the resuspension of particles deposited in the river bed. Rates of erosion are associated with climate, particularly the amount and intensity of rainfall, and can be modified by vegetative cover. Deforestation, or increased intensive agriculture results in large increases in erosion.

Although a general increase can usually be observed in suspended sediment concentration with increasing water discharge, it can be affected by a number of river basin processes. The relationship between discharge and suspended sediment for the River Exe, UK was discussed in section 4.3.3 and illustrated in Figure 4.4. Depending on the overall characteristics of the watershed, the peak in suspended sediment may, or may not, occur at the same time as the peak discharge.
6.2.4 In-stream velocities

In order to design effectively a water sampling programme, and to interpret river water monitoring results, some knowledge of the in-stream flows and velocity gradients is necessary. Within an even river channel, laminar flow occurs as indicated in Figure 6.7. Maximum velocities occur in the centre of the channel but are reduced to zero at the bank by frictional forces exerted by the shallow bank zone and the bank itself. The velocity gradient thus tends to force any influent waters from a tributary, industrial or municipal point source to the side of the river which they entered. A tributary entering a channel as illustrated in Figure 6.7 remains on the right hand side of the channel whilst laminar flow is maintained. Bends in the river, rapids or a waterfall induce mixing (bends by overturn and rapids and waterfalls by turbulent mixing). Concentration gradients, in the circumstances illustrated in Figure 6.7, follow the patterns indicated in the three cross-sections. Laminar flow, and the concentration gradients observed in the river sections, are normally maintained for less than one kilometre where perfect mixing occurs in small tributaries and turbulent rivers. Occasionally they are maintained for many hundreds of kilometres, as in the River Amazon downstream of the Rio Negro confluence. The implications for sampling cross-sections of rivers are discussed in section 6.6.4 and further details are available in UNESCO (1982).
Other examples of the velocity characteristics of rivers are given in Figure 6.8. The cross-sectional velocities are shown for a uniform cross-section (as in Figure 6.8A) or variations in cross-section (such as in Figure 6.8B) where major changes in depth occur across the profile. Figure 6.8C indicates the vertical velocity profile where bottom friction results in deceleration of the bottom water. The energy released by the deceleration is transferred into the movement of coarser sediment particles along the bed of the river. Figure 6.8D illustrates the overturn associated with a river bend. The cross-section (Figure 6.8E) shows the direction of circulation, but as forward motion is also maintained the resultant motion is roughly helical.

6.3. Chemical characteristics

At a given river station water quality depends on many factors, including: (i) the proportion of surface run-off and groundwater, (ii) reactions within the river system governed by internal processes, (iii) the mixing of water from tributaries of different quality (in the case of heterogeneous river basins), and (iv) inputs of pollutants.
6.3.1 Origins of elements carried by rivers

In the absence of any human impact the concentrations, relative proportions, and rates of transport of dissolved substances in rivers are highly variable from one place to another, depending on their sources, pathways and interactions with particulates. Particulate matter composition is discussed in detail in Chapter 4. Figure 6.9 shows the main sources of elements to rivers:

- **Chemical weathering** of surficial rocks (Figure 6.9, sources 1, 2 and 3). The most abundant rock types are shales (33.1 per cent of continental outcrops), granite and gneisses (20.8 per cent), sandstone (15.8 per cent), carbonate rocks (15.9 per cent) and basalts (4.1 per cent). Although rarely found at the earth surface (1.3 per cent), the evaporitic rocks, gypsum and rock salt, may greatly influence surface waters due to their very high solubility. Most chemical weathering reactions derive from the attack of minerals, mostly aluminosilicates, by carbonic acid (H$_2$O + CO$_2$). This leads to the formation of major cations (Ca$^{2+}$, Mg$^{2+}$, Na$^+$, K$^+$) and of dissolved silica (SiO$_2$) and bicarbonates (HCO$_3^-$). The less soluble trace elements (e.g. Fe, Al, Ti) remain in residual soil minerals (i.e. oxides and clay minerals). Weathering of carbonate rocks (about 15 per cent of rock outcrops at the earth’s surface) results in high concentrations of HCO$_3^-$, only half of which originates from carbonate minerals (the other half comes from...
atmospheric and soil CO₂). The sulphate anion is occasionally dominant when high proportions of pyrite (FeS) or gypsum (CaSO₄·2H₂O) occur. Chloride anion dominance is very seldom found in surface waters due to the scarcity of the NaCl mineral, but it may be important in coastal regions where it originates entirely from sea salt aerosols.

- **Atmospheric inputs** of natural origin (Figure 6.9, source 9). The amount of recycled oceanic aerosols (source 5: rich in Na⁺, Cl⁻, Mg²⁺, SO₄²⁻) falling on continents generally decreases exponentially from the coast inland, where the products of continental aeolian erosion (source 6: dust rich in Ca²⁺, HCO₃⁻, SO₄²⁻), of vegetation decay (source 7: rich in N species) and volcanic fall-out (source 4: HCl, H₂SO₄) may be dominant.

- **Leaching of organic soils** (Figure 6.9, source 8). This process generates nitrogen (NH₄⁺, NO₃⁻) and dissolved organic matter (dissolved organic carbon and nitrogen [DOC and DON]) in surface waters.

**Figure 6.9 Natural sources of elements to rivers.** 1, 2 and 3: Chemical weathering of surficial rocks; 4: Volcanic fallout; 5: Recycled oceanic aerosols; 6: Continental aeolian erosion; 7: Decay of vegetation; 8: Leaching of organic soils; 9: Atmospheric inputs

Anthropogenic activities can enhance natural processes, such as erosion and soil leaching, increase inputs of natural compounds such as mineral salts and inorganic fertilisers to the river system, and add synthetic compounds which are mostly organic and not found in nature, such as solvents, pesticides, aromatic hydrocarbons, etc. The additional materials arising from increases in natural processes follow the same pathways, and behave in the same way, as compounds arising from soil leaching, such as fertilisers and pesticides. However, most urban pollutants enter rivers as point-sources, usually as treated or untreated sewage effluents.
6.3.2 Natural concentrations in rivers

In any region not yet affected by human activity, the variability in natural water quality depends on the combination of the following environmental factors (Meybeck and Helmer, 1989):

- the occurrence of highly soluble or easily weathered minerals of which the order of weathering is halite > gypsum > calcite > dolomite > pyrite > olivine,

- the distance to the coastline,

- the precipitation/river run-off ratio, and

- the occurrence of peat bogs, wetlands and marshes which release large quantities of dissolved organic matter.

Other factors include the ambient temperature, thickness of weathered rocks, organic soil cover, etc.

Examples of the concentrations of ions and nutrients occurring in pristine waters are presented in Table 6.2. The geographic variability of natural running waters is surprising and as a result no “world average quality” can be used as a reference to check if a given river is polluted or not. Careful investigation of pristine water quality from elsewhere in the watershed should be made for comparison.

The natural geographic variation of selected dissolved constituents in rivers is given in Table 6.3 for pristine streams and major world rivers (highly polluted rivers in Europe and North America are not included). Within a given region of $10^3$ to $10^6$ km$^2$, the distribution of natural ionic contents between tributaries (areas $10^3$-$10^4$ km$^2$) may extend over several orders of magnitude, as for the Lower Amazon or the Mackenzie river tributaries (Figure 6.10). Internal processes which depend on the physical, chemical and biological conditions occurring in rivers, can also affect all components of water quality. Some major processes are listed in Table 6.4.

Dissolved trace element contents are very difficult to analyse correctly since samples are easily contaminated and analytical detection limits are sometimes higher than natural levels. The values given in Table 6.5 are the latest estimates found in the scientific literature for uncontaminated waters, resulting from utmost care in sampling, water treatment, and analysis. In routine surveys where adequate precautions are often not fully applied, the concentrations of dissolved trace elements may sometimes exceed (by one or even two orders of magnitude) the values reported in Table 6.5. Further information on the natural variations of river water quality can be found in Hem (1989) and in Berner and Berner (1987).
### Table 6.2 Geographic variability of dissolved major elements in pristine waters

|                        | Electrical conductivity (µS cm⁻¹) | pH | Sum of cations (µeq l⁻¹) | SiO₂ (mg l⁻¹) | Ca²⁺ (mg l⁻¹) | Mg²⁺ (mg l⁻¹) | Na⁺ (mg l⁻¹) | K⁺ (mg l⁻¹) | Cl⁻ (mg l⁻¹) | SO₄²⁻ (mg l⁻¹) | HCO₃⁻ (mg l⁻¹) | Notes |
|------------------------|----------------------------------|----|--------------------------|---------------|---------------|---------------|---------------|-------------|-------------|--------------|----------------|-------------|-------|
| **A. Pristine streams draining most common rock types (corrected for oceanic aerosols)** |                                  |    |                          |               |               |               |               |             |             |               |                 |       |
| Granite                | 35                               | 6.6| 166                      | 9.0           | 0.78          | 0.38          | 2.0           | 0.3         | 0           | 1.5           | 7.8           | 1     |
| Gneiss                 | 35                               | 6.6| 207                      | 7.8           | 1.2           | 0.69          | 1.8           | 0.4         | 0           | 2.7           | 8.3           | 1     |
| Volcanic               | 50                               | 7.2| 435                      | 12.0          | 3.1           | 2.0           | 2.4           | 0.55        | 0           | 0.5           | 25.9          | 1     |
| Sandstone              | 60                               | 6.8| 223                      | 9.0           | 1.8           | 0.75          | 1.2           | 0.82        | 0           | 4.5           | 7.6           | 1     |
| Shale                  |                                  |    |                          |               |               |               |               |             |             |               |                 |       |
| Carbonate              | 400                              | 7.9| 3,247                    | 6.0           | 51            | 7.8           | 0.8           | 0.51        | 0           | 4.1           | 195           | 1     |
| **B. Pristine streams draining rare rock types or in a rare geological formation** |                                  |    |                          |               |               |               |               |             |             |               |                 |       |
| Amazonian dear waters  | 5.7                              | 5.1| 111                      | 1.9           | tr            | 0.13          | 1.2           | 1.4         | 0.7         |              |               | 2     |
| Amazonian black waters | 29.1                             | 3.7| 212                      | 0.6           | tr            | 0.02          | 1.2           |             |             |              |               | 2     |
| Coal shale             |                                  |    |                          |               |               |               |               |             |             |               |                 |       |
| Salt rock              | 8.0                              | 9.0| 40,700                   | 7.0           | 87            | 121           | 600           | 10.2        | 14.8        | 1,400         | 652            | 2     |
| **C. Rivers influenced by evapotranspiration** |                                  |    |                          |               |               |               |               |             |             |               |                 |       |
| Horocallo R., Ethiopia | 2,230                            | 9.2| 21,800                   | 79            | 2.2           | 1.45          | 480           | 2.5         | 195         | 65           | 975            | 2     |
| **D. Rivers influenced by oceanic aerosols** |                                  |    |                          |               |               |               |               |             |             |               |                 |       |
| Clisson R., France     | 227                              | 6.2| 14.6                     | 6.4           | 4.8           | 22.0          | 2.6           | 40.0        | 5.8         | 32.9          |                 | 3     |
| tr Trace amount        |                                  |    |                          |               |               |               |               |             |             |               |                 |       |

1. Averages from a survey of 250 pristine streams in France (Meybeck, 1986) and from 75 sites world-wide, corrected for oceanic cyclic salts (Meybeck, 1987).

2. Compiled from various sources by Meybeck and Helmer, 1989.

3. Analyses not corrected for cyclic salt. 60% of salts originate from oceanic atmospheric inputs. River located in SW France, on sands (Meybeck, 1986).

Source: After Meybeck and Helmer, 1989
Table 6.3 Natural ranges of dissolved constituents in rivers

<table>
<thead>
<tr>
<th>Constituent</th>
<th>Streams (1-100 km²)</th>
<th>Rivers (100,000 km²)</th>
<th>Global average</th>
<th>MCNC</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Minimum (µeq l⁻¹)</td>
<td>Maximum (µeq l⁻¹)</td>
<td>Minimum (mg l⁻¹)</td>
<td>Maximum (mg l⁻¹)</td>
</tr>
<tr>
<td>SiO₂⁺ (µmole l⁻¹)</td>
<td>10</td>
<td>0.6</td>
<td>830</td>
<td>50</td>
</tr>
<tr>
<td>Ca²⁺</td>
<td>3</td>
<td>0.06</td>
<td>10,500</td>
<td>210</td>
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<td>Mg²⁺</td>
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<td>6,600</td>
<td>80</td>
</tr>
<tr>
<td>Na⁺</td>
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<td>0.06</td>
<td>15,000</td>
<td>350</td>
</tr>
<tr>
<td>K⁺</td>
<td>3</td>
<td>0.1</td>
<td>160</td>
<td>6.3</td>
</tr>
<tr>
<td>Cl⁻</td>
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<td>0.09</td>
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<td>530</td>
</tr>
<tr>
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<td>2.9</td>
<td>0.14</td>
<td>15,000</td>
<td>720</td>
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<tr>
<td>HCO₃⁻</td>
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<td>0</td>
<td>5,750</td>
<td>350</td>
</tr>
<tr>
<td>Sum of cations</td>
<td>45</td>
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<td>340</td>
<td>4,000</td>
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<tr>
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<td>4.7</td>
<td>8.5</td>
<td>6.2</td>
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</tr>
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<td>TSS</td>
<td>3</td>
<td>15,000</td>
<td>10</td>
<td>1,700</td>
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<tr>
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<tr>
<td>POC %</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>TOC</td>
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</tr>
<tr>
<td>N-NH₄⁺</td>
<td>0.005</td>
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<td>0.015</td>
<td></td>
</tr>
<tr>
<td>N-NO₃⁻</td>
<td>0.05</td>
<td>0.2</td>
<td>0.10</td>
<td></td>
</tr>
<tr>
<td>N organic</td>
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<td>1.0</td>
<td>0.26</td>
<td></td>
</tr>
<tr>
<td>P-PO₄³⁻</td>
<td>0.002</td>
<td>0.025</td>
<td>0.010</td>
<td></td>
</tr>
</tbody>
</table>

Streams: Distribution based on 75 unpolluted monolithological watersheds from all countries in which the rock type proportion is close to the estimated global proportion of Meybeck (1987), particularly for the most soluble rocks; oceanic cyclic salts have been grossly subtracted.

Rivers: These figures are derived from the discharge-weighted distribution of constituents in 60 major rivers (basic data in Meybeck, 1979) without any correction of oceanic cyclic salts.

Minimum and maximum values correspond to 2% and 98% of the distribution except for nutrients which represent 10% and 90%.

MCNC (most common natural concentrations) corresponding to the median value obtained for the same 60 major rivers as above.

TSS Total suspended solids
6.3.3 Variations of water quality with river discharge

Water quality variability depends on the hydrological regime of the river, i.e. the water discharge variability, the number of floods per year and their importance etc. (see section 6.2.2). During flood periods, water quality usually shows marked variations due to the different origins of the water: surface run-off, sub-surface run-off (i.e. water circulation within the soil layer), and groundwater discharge. Surface run-off is generally highly turbid and carries large amounts of total suspended solids, including particulate organic carbon (POC). Sub-surface run-off leaches dissolved organic carbon and nutrients (N and P) from soils, whereas groundwaters provide most of the elements resulting from rock weathering (SiO₂, Ca²⁺, Mg²⁺, Na⁺, K⁺).
The atmosphere is the source of most Cl\(^{-}\) and SO\(_4\)\(^{2-}\), together with some Na\(^{+}\), particularly in basins where there is no evaporitic rock or the mineral pyrite. Bicarbonate (HCO\(_3\)\(^{-}\)), the most common form of inorganic carbon found between pH 6 and 8.2, is derived partly from the dissolution of carbonate-bearing minerals and partly from CO\(_2\) in the soil. When the river basin does not bear any carbonate rocks, the HCO\(_3\)\(^{-}\) content is largely derived from soil CO\(_2\).

### Table 6.4 Some major internal processes regulating the concentrations of selected water quality variables in rivers

<table>
<thead>
<tr>
<th></th>
<th>Water turbulence</th>
<th>Evaporation</th>
<th>Adsorption on sediments(^1)</th>
<th>Primary production(^2)</th>
<th>Oxidation of organic matter</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>In the water column(^2)</td>
</tr>
<tr>
<td>pH</td>
<td></td>
<td>Increase</td>
<td>Increase</td>
<td>Decrease</td>
<td>Decrease</td>
</tr>
<tr>
<td>Electrical conductivity</td>
<td></td>
<td>Increase</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Calcium, bicarbonate</td>
<td>Precipitation</td>
<td>Precipitation</td>
<td>Precipitation</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sodium, chloride potassium, calcium sulphate</td>
<td>Increase</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nutrients</td>
<td>Volatilisation of NH(_3)</td>
<td>Decrease</td>
<td>Decrease (uptake)</td>
<td>Increase (release)</td>
<td>Decrease (denitrification) or increase (ammonification)</td>
</tr>
<tr>
<td>Dissolved O(_2)</td>
<td>Increase(^5)</td>
<td>Increase</td>
<td>Decrease</td>
<td>Decrease</td>
<td></td>
</tr>
<tr>
<td>Dissolved organic carbon</td>
<td>Decrease (foam formation and oxidation)</td>
<td>Decrease</td>
<td>Increase</td>
<td>Decrease</td>
<td></td>
</tr>
<tr>
<td>Dissolved metals</td>
<td></td>
<td>Decrease</td>
<td></td>
<td>Increase (desorption)</td>
<td></td>
</tr>
<tr>
<td>Organic micropollutants</td>
<td>Volatilisation</td>
<td>Decrease</td>
<td></td>
<td>Increase</td>
<td></td>
</tr>
</tbody>
</table>

\(^1\) During increased total suspended solids such as during floods

\(^2\) In natural river systems these processes are of minor importance in river channels, except in highly eutrophic rivers; these processes can exert major influences on water quality in lakes, reservoirs and impoundments
De-gassing of karstic waters

Due to pH increase

Re-aeration

Changes in discharge, when compared to the simultaneous changes in concentrations of various substances, are of great value in indicating the major sources of substances. This is illustrated by the curves of concentration versus discharge in Figure 6.11 A. Such curves may represent time scales varying from a single storm event to several years duration. Curve (1) shows a general decrease in concentration with discharge which implies increasing dilution of a substance introduced at a constant rate (e.g. major cations, possibly SiO₂, particularly when concentrations are high). This situation is also characteristic of point source discharges such as municipal sewage and many industrial point sources (see the example of PC₄³ in Figure 6.12). Curve (2) shows a limited increase in concentration generally linked to the flushing of soil constituents (e.g. organic matter, nitrogen species) during run-off. Curve (3) is basically the same as curve (2) but a fall off in concentration occurs at very high discharges indicating dilution of the soil run-off waters. Curve (4) shows an exponential increase in concentration which occurs with TSS and with all substances bound to particulate matter. The curve represents the increase in particulate matter due to sheet erosion and bed remobilisation. Substances bound to such particulates include phosphorus, metals and organic compounds, predominantly pesticides and herbicides. Curve (5) is the hysteresis loop observed as time is introduced as an additional parameter to the sediment discharge relationship shown in curve (4). Such patterns can be seen for TSS, DOC and sometimes NO₃. The peak in sediment concentration occurs at X (advanced) before the occurrence of the peak discharges at Z. Curve (6) indicates a water source to the river with a constant, or near constant, concentration (e.g. Cl⁻ in rainfall, groundwater in karstic regions or an outlet from a lake).

Table 6.5 World average values of trace elements carried in solution by major unpolluted rivers

<table>
<thead>
<tr>
<th>Element</th>
<th>Al (µg l⁻¹)</th>
<th>As (µg l⁻¹)</th>
<th>B (µg l⁻¹)</th>
<th>Cd (µg l⁻¹)</th>
<th>Cr (µg l⁻¹)</th>
<th>Co (µg l⁻¹)</th>
<th>Cu (µg l⁻¹)</th>
<th>F (µg l⁻¹)</th>
<th>Fe (µg l⁻¹)</th>
<th>Mn (µg l⁻¹)</th>
<th>Mo (µg l⁻¹)</th>
<th>Ni (µg l⁻¹)</th>
<th>Pb (µg l⁻¹)</th>
<th>Sb (µg l⁻¹)</th>
<th>Sr (µg l⁻¹)</th>
<th>V (µg l⁻¹)</th>
<th>Zn (µg l⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dissolved concentration</td>
<td>40</td>
<td>1.0</td>
<td>30</td>
<td>0.01</td>
<td>0.1</td>
<td>0.1</td>
<td>1.4</td>
<td>100</td>
<td>50</td>
<td>10</td>
<td>0.8</td>
<td>0.4</td>
<td>0.04</td>
<td>100</td>
<td>0.2</td>
<td></td>
<td></td>
</tr>
<tr>
<td>% of total in solution</td>
<td>0.13</td>
<td>25</td>
<td>40</td>
<td>2.5</td>
<td>0.25</td>
<td>2.5</td>
<td>20</td>
<td>0</td>
<td>2</td>
<td>30</td>
<td>4</td>
<td>0.5</td>
<td>45</td>
<td>68</td>
<td>1.4</td>
<td>0.2</td>
<td></td>
</tr>
</tbody>
</table>

¹ From Schiller and Boyle, 1985, 1987; Meybeck, 1988

² Modified from Meybeck and Helmer, 1989
Figure 6.11 A. Patterns of concentration C with water discharge Q in rivers; B. and C. The same patterns illustrated as synchronous (B.) and asynchronous (C.) variations in concentration and discharge over time during a single storm event. Points X and Z represent the maximum concentration and peak discharge respectively. For further details see text. (Modified from Meybeck et al., 1989)
The concentration discharge relationships (curves 1-6) indicated in Figure 6.11A are also illustrated as changes in concentration and discharge with time during a single storm event in Figures 6.11B and 6.11C. Simple relationships have been chosen to demonstrate their diagnostic capability in determining major sources of pollutants. Curve (1) in Figure 6.11A and B illustrates a simultaneous dilution effect and curves (2) and (4) illustrate simultaneous increases in concentration with discharge. Curves (3) and (5) in Figure 6.11A and C illustrate moderate and rapid increases in concentration prior to the peak discharge. A pollutant is, however, usually derived from a number of sources, resulting in a more complicated situation. This is illustrated for total PCBs (polychlorinated biphenyls) in the River Seine, France (Figure 6.12). Three general relationships can be observed between PCB concentration and discharge:

- a high coincidence with high concentrations (PCBs occurring with a high discharge in January and November),
- a moderate relationship where peak concentrations of PCBs relate to only moderate peaks in discharge (as in early January and June), and
- maximum discharge associated with low PCB concentrations in March, April and May.

This indicates two major sources of PCB to this river system: an upstream source in which the PCB became adsorbed by sediment particles, and a more localised source providing soluble PCB to the river system. The sediment related source is active in January and some sediment exhaustion probably occurs since the soluble source predominates during the high discharges in March, April and May. Sediment with
adsorbed PCB is re-accumulated in the river bed during summer slow flow and is reactivated by resuspension, thereby appearing as the major source in September and November.

Major sewage treatment plants release a constant flux of pollutants throughout the year which is diluted by the receiving river water. Figures 6.13A and 6.13B show the variation in orthophosphate with water discharge downstream of the Achères sewage outfall on the River Seine. Most of the wastewaters of Paris are discharged at Achères after primary or secondary treatment. The orthophosphate pattern shows a near-perfect dilution (Figure 6.13A, B) with a near-constant flux (Figure 6.13C). The weekly sampling frequency could be changed to a bimonthly, or even monthly, frequency if these relationships are taken into account.

Figure 6.13 A. Water discharge and orthophosphate concentrations in the River Seine from 1982 to 1986 downstream of the Achères sewage outfall; B and C. The orthophosphate shows a nearly perfect dilution with a nearly constant flux (Data courtesy of Service de la Navigation de la Seine, Rouen)

6.3.4 Temporal variations in water quality: trends and fluxes

The temporal distribution of concentrations can be determined either with a low sampling frequency over a long time period or with a high frequency over a minimum period of one
year (i.e. one hydrological cycle). When the sampling station represents a simple aquatic system, the data obtained are from a single statistical population which, in most cases, has either a normal (Gaussian) or log-normal distribution. This can be checked on special probability paper where the cumulative frequency curve is plotted (see Chapter 10, and Figure 6.10 for the use of such plots in spatial distributions). If a river is permanently polluted, the data distribution is also normal or log-normal but shows a general shift towards higher concentrations for most compounds or towards lower concentrations, as for dissolved O$_2$. In the case of intermittent pollution (e.g. seasonal anthropogenic activities, periodic releases of pollutants, accidents), the statistical distribution of measurement data tends to show a distinct change which characterises two different populations. However, the data should be broken down into seasonal aggregations because marked seasonal variations can occur in water quality variables which do not reflect intermittent pollution. An example of statistical distributions is given in Figure 6.14 for the Laita river in Brittany. Such distributions of data (see also Chapter 10) are also useful for checking the proportion of time that water is fit for various uses defined by water quality standards or criteria.

Long-term trends in water quality cannot usually be determined for less than ten years of monitoring. The longest records (over 100 years) in Europe are for the River Thames in the UK and the River Seine in France. As the errors in annual average concentrations in rivers are usually at least ± 20 per cent, only considerable changes in water quality can be shown over short periods, such as the reduction of TSS downstream of a major reservoir following dam closure. Comparisons of time distributions at 10 to 20 year intervals are useful for determining whether the water quality at a river station has changed. A description of methods for trend detection is given in Chapter 10. Although very tedious to establish, trends are essential for assessing the efficiency of clean-up measures in a river basin, or for determining the necessary actions to combat pollution.

Measurement of fluxes of chemical compounds (e.g. nutrients, micropollutants, mineral salts, organic matter) is sometimes a key objective in river assessments at international boundaries, in lake tributaries, or upstream of the estuarine zone. Theoretically, fluxes ($\Phi$) (in mass per unit time, usually t a$^{-1}$) are derived from the continuous measurements of both water discharge $Q$ (m$^3$ s$^{-1}$) and concentrations $C$ (mass per volume, usually mg l$^{-1}$) between time $t_1$ and $t_2$.

$$\Phi = \int_{t_1}^{t_2} C(t)Q(t)dt$$
Figure 6.14 Statistical time distributions of Mg$^{2+}$, Na$^+$, HCO$_3^-$, pH, NO$_3^-$ and O$_2$ in the Laita river, France. Results from monthly measurements over five years. (Data from Ministère de l’Environnement, Paris, 1976-1980 survey)

However, continuous determination of concentration is rarely done and it is usually necessary to rely on discrete water quality information obtained at fixed sampling periods. The water discharge, however, may be recorded continuously with a river stage recorder (limnigraph) and determined with an appropriate calibration curve for the sampling station. The actual concentrations occurring during the time between two chemical analyses must be extrapolated. For the fluxes of particulate pollutants (see Chapter 4), this can be done in two ways: by the constant concentration hypothesis or by the constant flux hypothesis:

- **Constant concentration:** The concentration $C_i$ measured at time $t_i$ is supposedly constant during a time interval $\delta t_i$ around the time of sampling. The flux $\psi_j$ of dissolved material discharged during this interval is: $\psi_j = Q_j C_j$ where $Q_j$ is the average water discharge during the interval $\delta t_j$. The total flux for the whole period is then: $\Phi = \Sigma \psi_j$. This assumption is particularly valid for compounds which increase in concentration during river floods (see Figure 6.11, curves 2, 3, 4 and 5), although the estimated flux is less than the real value.

- **Constant flux:** The flux, $\psi_i = Q_i C_i$, measured at the sampling time $t_i$, is supposedly constant during the time interval $\delta t_i$. The total flux for the whole period is then: $\Phi = \Sigma \psi_i$. This assumption is more valid for compounds which decrease in concentration during river floods (see Figure 6.11, curves 1 and 6), although the estimated flux is greater than the real value. Other methods have been proposed by Thomas (1986) and Walling and Webb (1988).
6.4. Biological characteristics

6.4.1 Factors affecting biological communities in running waters

Flowing waters are complex ecosystems consisting of different habitats (biotopes) and biotic communities (biocoenoses). The physical structure of the ecosystem may be broadly divided into: the water body and stream bed (aquatic zone), the water exchange zone (lentic zone and flood plain) and the environment influenced by the water (terrestrial zone) (see Figure 6.2). For the purposes of water quality assessment the aquatic zone is the most important. The three zones are characterised by specific hydrological features which directly, or indirectly, govern the biological communities that thrive there. The characteristics of the habitats vary from the head-waters to the sites of eventual discharge to receiving waters. Consequently, the biological communities also vary, not only from site to site, but along the length of the river.

For successful colonisation of the flowing water mass, living organisms have to adopt a variety of basic, life strategies, principally:

• to exhibit growth patterns and survival techniques which can withstand the relatively short retention times,

• to exploit “refuge” spaces or boundary layers, or

• to have the ability to swim against the prevailing currents.

For small organisms unable to swim against the current, adaptations include a flattened or spindle-shaped body, as well as adhesive devices and the occupation of the spaces with little or no water flow. In this way the organisms are not carried away by the current and can benefit from the flowing water which provides continuous, rapid exchange of oxygen and nutrients.

In considering a flowing water ecosystem, two important aspects must be taken into account:

• Continuous water flow allows any input, such as an effluent, to have an effect locally as well as along the downstream course of the river.

• As river water is usually retained within the watercourse for relatively short periods (days to a few weeks) before being replaced from other sources, time dependent processes, such as growth or degradation, have only a limited time period within which they can show their effects.

A number of physico-chemical characteristics are of particular importance in determining the biological nature of river systems through their modification of suitable habitats. These characteristics are: flow rate, erosion and deposition, substrate nature, light, temperature and oxygen.
Flow rate

The velocity of water within a river has direct and indirect effects on the biota. It supports or carries organisms, determines the physical structure of the stream bed, and has considerable influence on surface exchange of gases. The roughness of the river bottom, as well as the flow pattern arising from it, are important for the formation of habitats in which organisms can survive (Figure 6.15). In most rivers, discharges vary seasonally, imposing seasonal changes on biological communities.

As flowing water allows turbulent support of a wide variety of types and sizes of particulate materials, the penetration of light into the water is often highly dependent on the stream velocity characteristics. This then has a direct bearing on the amount, type and distribution of the photosynthetic organisms which can colonise the various habitats.

Erosion and deposition

Rivers are subject to continuous change through erosion and deposition. In normal conditions, this can lead to the displacement of the stream bed and the channel line. Such effects may also be artificially magnified by human activities in the terrestrial zone (river bank modification for flood prevention) or by canalisation. Erosion which is intensified by human activity leads to the loss of habitats and a reduction in biological communities in the affected reaches of the river.

Occasional movement and displacement of deposited sediments and rocks are normal processes in flowing waters, and have little impact on biological communities. Permanent displacement on a large scale (particularly in areas of high erosion) frequently occurs in fast flowing rivers, especially in tropical countries, and tends to prevent colonisation by organisms.

Substrate

The substrates available for colonisation by biota vary considerably in rivers, such as solid rock, stones, gravel, sand or sludge. Roots, dead wood, as well as submerged spermatophytes, mosses, filamentous algae, reeds and floating leaf plants also form natural substrates. Artificial structures, such as concrete walls, wood and metal sheet piling, may also provide suitable, but limited, habitats. Fine-grained substrates (mud and sand) are preferentially colonised by diatoms, blue-green algae and higher plants, as well as by certain species of worm, insects and other animals. Rocky bottomed streams do not usually support the growth of many macrophyte species but the surfaces of the rocks may provide suitable habitats for attached algae, such as certain diatoms.
Figure 6.15 Water flow over a stony stream bed; the pockets of still water behind stones provide suitable habitats for colonisation by aquatic organisms

Mobile animals generally prefer the sides of substrates which are sheltered from light and the water current (Figure 6.15). Consequently, the spaces between and beneath stones are particularly suitable habitats, as are moss layers and the standing crop of water plants. Sheltered spaces with slow flow conditions offer favourable living conditions to many invertebrates and also provide spawning grounds for fish. The pore spaces below the stream bed (hyporheic interstitial - see Figure 6.2) are also biologically important habitats, providing refuges for the early development stages of invertebrates and fish and a suitable medium for the self-purification processes carried out by micro-organisms.

Light

Light is required for photosynthesis by all river primary producers, i.e. the algae and macrophytes. Sub-surface light is usually exponentially absorbed in its downwards passage through the water column. The depth of the euphotic zone (i.e. the zone with sufficient light to support photosynthetic activity) in rivers is highly dependent on the water colour and the amount of suspended sediment present. Frequently, river and stream water which is very clear is fairly shallow allowing sufficient light penetration to support benthic algae or attached macrophytes. In the lower reaches of a river where the water is deeper, or more turbid, the euphotic zone may be wholly contained within the main water mass. Thus, except for the littoral shallows, plant colonisation of the sediments is not possible.

Temperature

Water temperature influences the rate of physiological processes of organisms, such as the microbial respiration which is responsible for much of the self-purification that occurs in water bodies. Higher temperatures support faster growth rates and enable some biota to attain significant populations. Under natural conditions the temperature of running water varies between 0 °C and 30 °C. Higher temperatures (> 40 °C) usually only occur in volcanic waters and hot springs. In running water, the temperature normally increases gradually from the source of the river to its mouth. Cooling waters discharged to rivers, e.g. from industrial activities or from power generation, can lead to higher than normal water temperatures. These increased temperatures cause problems for sensitive organisms due to the increased oxygen demand (lowering oxygen saturation) and increased level of toxicity of harmful substances. They are sometimes also responsible for fish kills.
Oxygen

Oxygen is one of the most important factors for water quality and the associated aquatic life. Oxygen deficiency, even if it occurs only occasionally and for short periods, leads to a rapid decrease in the number of aquatic animals present, particularly the clean water species which depend on high oxygen levels, as well as most fish. In slow-flowing or impounded rivers, the effects of eutrophication (nutrient enrichment) can lead to deoxygenation of the sediments and possible remobilisation of nutrients and toxic trace elements, particularly from the sediments.

6.4.2 Pelagic communities

The pelagic communities are those swimming or floating organisms associated with the free water in the aquatic zone of the river, e.g. plankton and fish.

The phytoplankton of rivers and streams are only able to attain obvious populations when their growth rates are such that sufficient population doubling times can be attained within the retention periods of the watercourses. Increased light, higher temperature and lower turbidity, together with reduced velocity which provides longer watercourse retention times, tend to promote greater phytoplankton growth rates. River management practices such as abstraction or weir controls can lead to increased phytoplankton growth rates. Phytoplankton are particularly sparse, or absent altogether, in small streams and more free-flowing rivers, particularly where there is little or no natural, or artificial, input of nutrients. In many tropical and subtropical rivers, phytoplankton communities do not reach high densities due to the very high levels of turbidity caused by suspended solids arising from land-based erosion processes. Submerged vegetation is, therefore, also rare.

Zooplankton are small, to microscopic, animals which feed on primary producers or their products, or on other zooplankton. The juvenile stages of larger zooplankton (> 1 mm) may last for several days and, therefore, substantial populations can only be produced in rivers with very low velocities and warm water. However, smaller zooplankton such as rotifers can attain quite large populations.

Fish have been able to exploit all physically accessible river habitats. Their eggs often adhere to stones or weeds, or may be deliberately placed in specially constructed refuges. Consequently, highly specific physical conditions are necessary in a river for successful fish breeding. Fish communities are, therefore, sensitive to modification of the river regime (velocity, erosion, etc.) as well as to the input of toxic substances.

Migratory fish which return from the sea to spawning grounds in upstream stretches of rivers and their tributaries can be prevented from reaching their spawning grounds by physical barriers or chemical barriers along their migratory route. Physical barriers consist of weirs, locks and dams and chemical barriers are stretches of highly toxic or anoxic water in the river. The removal, or bypassing, of barriers to fish migration is fundamental to the restoration of water quality to a level which is acceptable by naturalists, fishermen, and the general public. Examples of schemes to restore self-sustaining populations of migratory salmonid fish can be found from several countries (e.g. Canada, UK, Japan) as well as for international rivers, such as the Rhine (IKSR,
1987). Such schemes provide visible evidence to the general public that river water quality has been improved.

6.4.3 Benthic and attached communities

The benthic communities of rivers and streams consist of those organisms which grow in, on, or otherwise in association with various bottom substrates. These communities are frequently used to assess changes in river water quality (see Chapter 5). As benthic organisms have limited mobility, their presence or absence is most likely to be associated with changes in their habitat or environmental conditions. The zoobenthos are primarily the invertebrate animals which live on, or are associated with, the river bed. They exhibit a very wide diversity of form, tolerance to environmental conditions and adaptation to survival in the different habitats of a river. A large proportion of the particulate organic carbon which enters a river, either as allochthonous or as autochthonous material, is processed by the zoobenthos.

Periphyton are microscopic plants which usually occur in quite thin layers on stones, rocks, sand and mud (epipelic algae). Epiphytic algae grow attached to the surfaces of higher plant stems, branches and leaves. These organisms are not easily swept away and their local nutrient environment is continuously replenished. In turbid conditions, caused by suspended solids or a phytoplankton bloom, algae on or near the bottom may not receive adequate light for photosynthesis and growth.

Macrophytes are plants which grow attached to, or rooted in, the substrate. The macrophyte species composition and abundance of established streams and rivers is usually fairly stable, but can show seasonal and annual differences. Some remain wholly submerged, whereas others produce surface-floating or truly aerial components. Since macrophytes require certain conditions of light penetration and nutrient availability, changes in the species composition and abundance can also be indicative of changes in water quality, especially eutrophication, or the physical characteristics of the river bed. Macrophytes can provide important refuges for small invertebrates, fish eggs and fry.

6.4.4 River zonation

Characteristic zones may be recognised in rivers and streams according to aspects of the habitats or biotic communities present, and the biological processes which occur along the length of the water course (Hawkes, 1975).

The aquatic zone of a river system is normally permanently submerged, and the associated communities are unable to withstand desiccation. The lentic zone and flood plain of a stream are the areas between the mean low water zone, e.g. the zone where reeds grow, and the mean high water limit. As a result, this area is subject to frequent, recurring fluctuations in water level. In large rivers the lentic zone may be very large and many metres wide, but in smaller rivers and streams it can be rather fragmented as the banks tend to be steep. As a result of increased erosion caused by human activity, particularly in deforested tropical areas, rivers and streams may become so deep that they cannot develop an active lentic zone.

Above the lentic zone and flood plain, at the mean high water level is the terrestrial zone. The lower limit of this zone is often indicated by the visible growth of small trees and
bushes. The terrestrial zone is usually considered as part of the alluvial valley, at least from the limit to which the valley floods (the recent alluvial plain). There is a close inter-relationship between a stream and its valley, especially with old channels, backwaters, depressions and flood channels. In dry periods the water quality of these may be very different from the main river due to groundwater inputs, anoxic conditions, denitrification, etc. During floods, river water flows through these water bodies allowing the exchange of water, substrates and organisms. The soil water and nutrient budgets of the flood area are usually characterised by the river water, and the nature and shape of the soil surface is changed by erosion and siltation. This transitional zone (ecotone) between the river and the land is currently the subject of much scientific investigation. It is becoming increasingly appreciated that there is an ecological continuum from the aquatic zone to the flood plain and that this area provides the basis for self-purification within the ecosystem (Amoros and Petts, 1993).

In some flowing waters the natural secondary production (i.e. the biomass of aquatic invertebrates and fish) depends less on the primary production in the water itself than on the primary production of the surrounding terrestrial zone. This primary production arises from plants (leaves, roots, flowers and fruits) and also animals, e.g. insects, entering the water from the air, the river banks or flood zone, providing a significant source of nutrients for fish.

**Habitat zonation**

It is possible to recognise a number of zones based on a variety of habitats, from source to receiving waters. The main habitat classification is based on the erosion or deposition characteristics of the water stretches (Table 6.6). As discussed above, the interaction of erosion and deposition determines the dominant particle size within the habitat. The combinations of dominant particle size and associated organic material support typical assemblages of dominant primary, secondary (macroinvertebrate) and tertiary (fish) producers (Table 6.6). Within the macroinvertebrate communities, further characteristic groups can be identified according to their principal feeding mechanism (i.e. grazer, shredder, collector or predator).

**Community zonation**

In considering the biological communities which may be found in a river reach, fish have often been used to characterise zones. As fish are widespread, and have a specific ability to exploit many habitats, the species or genera that occur can be indicative of the water quality characteristics of the zone, particularly in association with other typical assemblages of organisms (Figure 6.16). The preferred habitats for fish may depend on such factors as suitable breeding sites (e.g. gravelly substrate, dense macrophyte growth or rapid flow), minimum oxygen concentrations or appropriate food supply. Such factors are usually typical of particular reaches of a river (see Table 6.6) and, therefore, habitat and community zonation are closely linked.

**Ecological zonation**

Zones can also be determined by the ratio of production (P) to degradation, as indicated by respiration (R) (i.e. the P:R ratio) together with the associated community structure of the macroinvertebrate populations (Figure 6.16). This can be a useful approach where
fish are sparse or absent. The upper zone is usually associated with a ratio: P/R < 1, and is dominated by organisms relying on filtering suspended matter or grazing on allochthonous material such as leaf detritus. The intermediate zone usually has a ratio: P/R > 1, and a dominance of organisms filtering suspended material and grazing on attached algae. The lower zone is characterised by a ratio: P/R < 1, and a dominance of organisms filtering suspended material.

Table 6.6 An example of river zonation based on the physical characteristics of riverine habitats and the major groups of organisms (northern temperate representatives) associated with each zone

<table>
<thead>
<tr>
<th>Habitat type</th>
<th>Dominant particle size</th>
<th>Primary producers</th>
<th>Grazers (scrapers)</th>
<th>Shredders (large particle detritivores)</th>
<th>Collectors (fine particle detritivores)</th>
<th>Predators</th>
<th>Fish</th>
</tr>
</thead>
<tbody>
<tr>
<td>Erosional</td>
<td>Coarse (&gt; 16 mm)</td>
<td>Diatoms, mosses</td>
<td>Mollusca (Ancylliae)</td>
<td>Plecoptera (Nemouridae, Pteronarcidi e, Peltoperlaid e, Tipulidae)</td>
<td>Ephemeroptera (Heptageniidae, Bae tidae, Siphlonuridae)</td>
<td>Plecoptera (Perlidae)</td>
<td>Plecoptera (Perlidae)</td>
</tr>
<tr>
<td>Intermediate</td>
<td>Medium (&gt; 1 mm &lt; 16 mm)</td>
<td>Green algae, aquatic plants (e.g. Heteranthera, narrow-leaved Potamogeton)</td>
<td>Mollusca (Sphaeridae, Pleuroceridae, Planorbiidae)</td>
<td>Trichoptera (Limnephilidae e.g. Pycnopsych e)</td>
<td>Ephemeroptera (Ephemerida)</td>
<td>Odonata (Corduligasteri dae, Petalidae)</td>
<td>Plecoptera (Perlidae)</td>
</tr>
<tr>
<td>Depositinal</td>
<td>Fine (&lt; 1 mm)</td>
<td>Aquatic plants (e.g. Elodea)</td>
<td>Mollusca (Physidae, Unionidae)</td>
<td>Trichoptera (Limnephilidae e.g. Platycrichtopus)</td>
<td>Oligochaeta Ephemeroptera (Hexagenia, Caenidae)</td>
<td>Diptera (Chironomidae, Chironominae)</td>
<td>Odonata (Gomphidae, Agrionidae)</td>
</tr>
</tbody>
</table>
6.5. Major water quality issues in rivers

6.5.1 Changes in physical characteristics

Temperature, turbidity and TSS in rivers can be greatly affected by human activities such as agriculture, deforestation and the use of water for cooling (see Table 6.7). However, in certain circumstances the same activities may have little effect in rivers. For example, tropical rivers may not be greatly affected by thermal wastes. In regions of high erosion (e.g. steep slopes, heavy rainfall, highly erodible rocks) where natural TSS already exceeds 1 g l\(^{-1}\) (see Chapter 4), intensive agriculture aggravates the natural erosion rates. The turbidity of many sub-arctic rivers, as well as some of those in the wet-tropics (e.g. Zaire and Amazon basins), may be naturally very high due to the colour of dissolved humic substances.
Table 6.7 Major human activities affecting the physical characteristics of rivers

<table>
<thead>
<tr>
<th>Activity</th>
<th>Temperature</th>
<th>Turbidity</th>
<th>Total suspended solids</th>
</tr>
</thead>
<tbody>
<tr>
<td>Damming</td>
<td>- to ++</td>
<td>-</td>
<td>- - -</td>
</tr>
<tr>
<td>Cooling water discharge</td>
<td>+ +</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Domestic sewage discharge</td>
<td>+</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Industrial wastewater discharge</td>
<td>+</td>
<td>+ +</td>
<td>+ + +</td>
</tr>
<tr>
<td>Intensive agriculture</td>
<td>+</td>
<td>+ + +</td>
<td></td>
</tr>
<tr>
<td>Navigation</td>
<td></td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Dredging</td>
<td>+</td>
<td>+ +</td>
<td></td>
</tr>
</tbody>
</table>

+ to + + + Slight to severe increase
- to - - - Slight to severe decrease

1 Depending on the depth of the water outlet with respect to the thermocline

6.5.2 Faecal contamination

Faecal contamination is still the primary water quality issue in rivers, especially in many developing countries where human and animal wastes are not yet adequately collected and treated. Although this applies to both rural and urban areas, the situation is probably more critical in fast-growing cities where the population growth rate still far exceeds the rate of development of wastewater collection and treatment facilities (Meybeck et al., 1989). As a result faecal coliform bacteria can be found in numbers of $10^6$ per 100 ml for some rivers passing through cities such as Djakarta, New Delhi and others.

Where active collection and treatment of wastewater has been carried out since the late 1960s, counts of faecal coliform bacteria have stabilised or even fallen. Levels are now commonly between 100 and 10,000 coliforms per 100 ml.

6.5.3 Organic matter

The release into rivers of untreated domestic or industrial wastes high in organic matter results in a marked decline in oxygen concentration (sometimes resulting in anoxia) and a release of ammonia and nitrite downstream of the effluent input (Figure 6.17). The effects on the river are directly linked to the ratio of effluent load to river water discharge. The most obvious effect of organic matter along the length of the river is the “oxygen-sag curve” (Figure 6.17A) which can be observed from a few kilometres to 100 km downstream of the input. The eventual recovery in oxygen concentrations is enhanced by high water turbulence. Some industrial activities (e.g. pulp and paper production, palm oil extraction and sugar beet processing) may produce wastewaters with BOD₅ and COD values exceeding 1,000 mg l⁻¹. When this wastewater is discharged into a river, oxygen can be completely depleted as in the Laita river (Figure 6.14).
When monitoring for the effects of organic matter pollution, stations should be located in the middle of the oxygen-sag curve (if the worst conditions are being studied) or at the beginning of the recovery zone, depending on the objectives of the programme. During preliminary surveys a complete longitudinal profile incorporating various hydrological features is necessary in order to choose the location of permanent monitoring stations.

6.5.4 River eutrophication

During the 1950s and 1960s, eutrophication (nutrient enrichment leading to increased primary production) was observed mostly in lakes and reservoirs. The increasing levels of phosphates and nitrates entering rivers, particularly in developed countries, have been largely responsible for eutrophication occurring in running waters since the 1970s. In small rivers (i.e. stream orders 3, 4 and 5, see section 6.2.1), eutrophication promotes macrophyte development, whereas in large rivers phytoplankton are usually more common than macrophytes. In such situations the chlorophyll levels may reach extremely high values (up to 200 mg m\(^{-3}\)) as in the River Loire, France or the River Rhine in Germany (Figure 6.18). Eutrophication in river systems can also be caused by the construction of reservoirs and locks (used for navigation), both of which produce a marked decrease in flow velocities within the river (see Chapter 8). Table 6.8 summarises the effects of eutrophication in different types of running waters.
Eutrophication can result in marked variations in dissolved oxygen and pH in rivers during the day and night. During daylight, primary production (P) far exceeds the bacterial decomposition of algal detritus (R), and O$_2$ over-saturation may reach 200 per cent or more, with pH values in excess of 10 during the early afternoon. During the night, this pattern is reversed and O$_2$ levels may fall to 50 per cent saturation and the pH may fall below 8.5 (Figure 6.19). When such rivers also receive organic wastes, the diel (day and night) cycle still exists (Figure 6.20), but the average O$_2$ saturation is much lower and the peak O$_2$ level may not reach 100 per cent saturation. When respiration levels become greater than the primary production (i.e. R > P) in the downstream reaches of rivers, or in their estuaries, the O$_2$ concentration can decline dramatically. Occasionally this can result in total anoxia, as in some turbid estuaries during the summer period.

Diel variations in water quality cause major problems for monitoring and assessment of eutrophic rivers. Sampling at a fixed time of the day can lead to a systematic bias in recorded O$_2$ and pH levels. Although chlorophyll and nutrients may also show some fluctuations in concentrations, these are generally within 20 per cent of the daily mean.
Table 6.8 The effects of eutrophication in running waters

<table>
<thead>
<tr>
<th>Type of water</th>
<th>Eutrophication effects</th>
</tr>
</thead>
<tbody>
<tr>
<td>Headwaters of streams in the shade</td>
<td>None</td>
</tr>
<tr>
<td>Headwaters and streams exposed to sun</td>
<td>Promotion of macrophytes or periphyton growth including filamentous algae</td>
</tr>
<tr>
<td>Medium-sized rivers</td>
<td>Promotion of periphyton and/or macrophyte growth</td>
</tr>
<tr>
<td>Major rivers</td>
<td>Plankton growth, macrophyte growth</td>
</tr>
<tr>
<td>Locks on medium-sized rivers</td>
<td>Large increases in plankton populations and floating macrophyte growth</td>
</tr>
</tbody>
</table>

The construction of major dams which result in impoundments with water residence times greater than one week may lead to severe eutrophication if the nutrient loads are high enough to support algal growth. Monitoring and assessment of impoundments is discussed in Chapter 8.

1 Average width < 1 m

2 Average width > 1 m < 20 m; average depth < 2m

3 Average width > 20 m; average depth > 2 m

The changes in river water quality caused by eutrophication can be a major cause of stress to fish due to the release (at high pH) of gaseous NH₃, which is highly toxic to fish. In slow flowing, eutrophic rivers excessive phytoplankton growth can lead to problems for direct drinking water source intakes and any subsequent treatment processes. A rough estimate of the phytoplankton organic carbon (mg l⁻¹) can be obtained from a measurement of total pigments (i.e. chlorophyll a + phaeopigments (degraded pigments)). The minimum organic carbon present (mg l⁻¹) is approximately equal to 30 times the total pigments (mg l⁻¹), although this relationship has only been tested on western European rivers (Dessery et al., 1984).

6.5.5 Salinisation

Increased mineral salts in rivers may arise from several sources: (i) release of mining wastewaters as in the Rhine basin (potash mines and salt mines) or as in the Vistula, Poland (salt mines), (ii) certain industrial wastewaters (as in Figure 6.14), and (iii) increased evaporation and evapotranspiration in the river basin (mainly in arid and sub-arid regions) resulting from reservoir construction, irrigation returns, etc.

Industrial and mining wastes result in increases in specific ions only, such as Cl⁻ and Na⁺ from potash and salt mines, SO₄²⁻ from iron and coal mine wastes, Na⁺ and CO₃²⁻ from some industrial wastes. However, evaporation affects all ions and as calcium carbonate reaches saturation levels, calcium sulphate-rich waters or sodium chloride-rich waters are produced. Ion content (principally Ca²⁺, Na⁺, Cl⁻, SO₄²⁻) can also be affected by other human activities such as domestic wastewater inputs, atmospheric pollution, the use of de-icing salts, fertiliser run-off, etc. However, the resulting increases in major ions are much less than those resulting from the three major causes of salinisation mentioned.
above. The changes in ionic contents and the ionic ratio of waters are very often linked to pH changes. Mine wastewaters are generally very acidic (pH ≤ 3), whereas industrial wastes may be basic (Figure 6.14) or acidic. Salinisation resulting from evaporation usually leads to more basic pH levels (see Table 6.2 where the pH reaches 9.2 in an arid region of Ethiopia).

Figure 6.19 Theoretical variations in O₂ and pH associated with algal production in a eutrophic river (P production; R respiration)
Figure 6.20 Modification of the diel oxygen cycle at two sites on the River Saar by the presence of organic matter pollution: A. Güdingen (unpolluted); B. Völklingen (polluted) (Based on Müller and Kirchesch, 1980)

As electrical conductivity is controlled by the major ion contents, continuous monitoring may be carried out using a conductivity meter linked to a recording device. Results should be checked occasionally for the relationship between the ion concentrations and the conductivity (see Figure 10.14).

Longitudinal profiles of chloride in rivers can help to determine the cause of salinisation such as for the Rhine (Figure 6.21) whereas long-term trends of concentration or fluxes assessed by carrying out measurements for over a decade or more can illustrate the results of legislative control of wastewater discharges and/or other remedial measures (Figure 6.22).

6.5.6 Acidification

Acidification can occur in running waters as a result of: (i) direct inputs of acidic wastewaters from mining or from specific industries, either as point sources (e.g. sewers) or diffuse sources (e.g. leaching of mine tailings), and (ii) indirect inputs through acidic atmospheric deposition, mainly as nitric and sulphuric acids resulting mostly from motor exhausts and fossil fuel combustion. In the latter case, acidification of surface waters may only take place if the buffering capacity of the river basin soil is very low. Low buffering capacity mainly occurs in areas of non-carbonate detrital rocks, such as sandstones, and of crystalline rocks such as granites and gneisses.

Point sources of acidic effluents to rivers may result in a substantial change in water quality downstream of the acid source. The extent of this kind of pollution is best monitored along longitudinal river profiles. Diffuse (i.e. atmospheric) acid inputs may
occur over extended regions (up to \(10^4\) km\(^2\)), sometimes located far downwind of the pollutant sources (100-1,000 km), such as major cities, major smelters, refineries, coal-burning power plants, etc. In colder regions where snow melt has a significant hydrological influence, the accumulated acidic deposition in the snow may be released when it melts. This can cause a sudden acid pulse which may be more than one pH unit lower than normal (Jones and Bisson, 1984).

Figure 6.21 Chloride concentrations along the River Rhine showing the downstream influence of the Alsace potash mines, 1971 (Data from Commission Internationale pour la Protection du Rhin Centre la Pollution, Koblenz)

![Chloride concentrations along the River Rhine](image)

Figure 6.22 Long-term trends in chloride flux in the Rhine at Kleve-Bimmen (Data from Landesamt für Wasser und Abfall NRW, Germany)

![Long-term trends in chloride flux](image)

A particular problem associated with acidification is the solubilisation of some metals, particularly of \(\text{Al}^{3+}\), when the pH falls below 4.5. The resultant increased metal
concentrations can be toxic to fish (see also Chapter 7), and also render the water unsuitable for other uses.

The assessment of diffuse source acidification in rivers must start with a precise inventory of the sensitive area and of the potential atmospheric pollutant sources. The actual river monitoring programme must be combined with the monitoring of atmospheric deposition and should also include specific measurements, i.e. $\text{Al}^{3+}$, other selected metals, acid neutralising capacity (ANC) (see Chapters 3 and 7) and dissolved organic carbon. In the area affected by acidification, long-term hydrological trends and specific events, such as snow melt, should also be thoroughly investigated.

Pristine areas where no direct sources of pollution occur, such as headwater streams and lakes, are generally the best places to locate monitoring stations for acidification, although they may be difficult to reach. Ideally a comprehensive monitoring network should be based on a few streams sampled intensively throughout the year, as well as a great number of sites sampled only a few times a year to check the spatial distribution of acidification. Past records of acidification may be obtained through the study of diatom species distributions especially in sediment cores (see Chapters 4 and 7) and biological methods can be used to determine an index of acidification (see Chapter 5).

6.5.7 Trace elements

Trace element pollution results from various sources, mostly: (i) industrial wastewaters such as mercury from chlor-alkali plants, (ii) mining and smelter wastes, such as arsenic and cadmium, (iii) urban run-off, particularly lead, (iv) agricultural run-off (where copper is still used as a pesticide), (v) atmospheric deposition, and (vi) leaching from solid waste dumps.

In surface waters, at normal pH and redox conditions, most trace elements are readily adsorbed onto the particulate matter (see Table 4.1). Consequently, the actual dissolved element concentrations are very low. The monitoring of trace elements on a routine basis is very difficult (see Chapter 3). Ambient air is often highly contaminated with many pollutants, particularly lead. Therefore, as a result of contamination during sample handling, many of the existing routine measurements of dissolved trace elements give much higher values (10 to 100 times higher) than measurements made by specialised sampling and analysis programmes (Meybeck and Helmer, 1989). Unless large financial resources are available for materials and training, the routine monitoring of dissolved trace elements is not recommended. Instead, analysis should be made on particulate matter samples, either suspended or deposited (Chapter 4), or on biological samples (Chapter 5).

The deposition of sediments with adsorbed contaminants provides a useful mechanism for the rapid evaluation of the distribution and origins of pollutants of low solubility in a river system. However, care must be taken to sample the fine bottom sediments which would have been recently accumulated and to take into account the variability of particle size in the interpretation of the data (see Chapter 4). Only rarely does long term accumulation allow the determination of the history of pollution in a river. An example from the River Rhine is given in Figure 6.23. Another example of the distribution of metals in river bed sediments is given in Figure 6.24 which shows the longitudinal profile of sediment contamination along the length of the Rio Paraiba Do Sul which runs
through an industrialised regions of Brazil and which receives pollutants from three different states. This river is also the source of drinking water for the major urban centre of Rio de Janeiro.

Figure 6.23 Accumulation of inorganic and organic pollutants in sediments taken from the mouth of the Old Rhine river (Based on Irion, 1982)

6.5.8 Nitrate pollution in rivers

Nitrate concentrations in some rivers of western Europe are approaching the World Health Organization (WHO) drinking water guideline value of 50 mg l\(^{-1}\) NO\(_3\) (Meybeck et al., 1989). Urban wastewaters and some industrial wastes are major sources of nitrate and nitrite. However, in regions with intensive agriculture, the use of nitrogen fertilisers and discharge of waste-waters from the intensive indoor rearing of livestock can be the most significant sources. Heavy rain falling on exposed soil can cause substantial leaching of nitrate, some of which goes directly into rivers, but most of which percolates into the groundwater from where it may eventually reach the rivers if no natural denitrification occurs (see Chapter 9).
Assessment of trends in nitrate concentrations should be undertaken on a long-term basis (i.e. frequent sampling for more than 10 years’ duration). However, in rivers already affected by organic wastes, causing a reduction in dissolved O₂ concentrations, denitrification may occur in the river bed sediments releasing N₂ to the atmosphere. Specific studies to quantify this process involve nitrogen budgets in river stretches and/or studies of interstitial waters from the river bed. However, reduction of NO₃⁻ to N₂ does not usually result in a significant reduction of the NO₃ load in rivers.

6.5.9 Organic micropollutants

Organic micropollutants (mostly synthetic chemicals manufactured artificially) are becoming a critical water quality issue in developed and developing countries. They enter rivers: (i) as point sources directly from sewers and effluent discharges (domestic, urban and industrial sources), (ii) as diffuse sources from the leaching of solid and liquid waste dumps or agricultural land run-off, or (iii) indirectly through long-range atmospheric transport and deposition. As an example, PCBs have been found in very remote sites of North America such as in the Isle Royale National Park in the middle of Lake Superior, where no human activity occurs. In some developing countries, agriculture is a major source of new chemical pollutants to rivers, such as pesticides. The approach to monitoring these substances depends mostly on their properties, i.e. volatility, water solubility, solubility in lipids, photodegradation, biodegradation, bioaccumulation, etc. (see Table 2.4 for some general guidelines on the most appropriate media for the monitoring of various types of organic micropollutants). An example of the monitoring of PCBs is given in Figure 6.12 for the River Seine at Paris.
where PCBs were monitored bimonthly in connection with TSS and POC monitoring (see also Figure 4.12).

**Figure 6.25 PCBs in the deposited sediments of the major tributaries of Lake Geneva (Based on Corvi et al., 1986)**

Another example of the use of deposited sediment is given in Figure 6.25 showing sediment-bound PCBs at the mouths of the rivers draining into Lake Geneva. Twenty-two rivers were analysed and the results are presented in five concentration class ranges. Of the three major rivers (Rhône, Dranse and Venoge), only the Venoge showed moderately high concentrations. The most contaminated sediments occurred at the mouths of the smaller tributaries. Care is needed to avoid misinterpreting the effect on the lake from the schematic data presented in Figure 6.25. The map indicates the rivers with the most polluted sediment (i.e. the highest concentrations of PCBs). However, the impact on the lake is determined by fluxes or loads, i.e. the total quantity of sediment delivered from any river multiplied by its pollutant concentration (see Chapter 7).
Table 6.9 Ecological impacts of direct or indirect modifications of the river bed

<table>
<thead>
<tr>
<th>Activity/modification</th>
<th>Effects</th>
</tr>
</thead>
</table>
| Construction of locks         | Enhancement of eutrophication  
Partial storage of fine sediments may result in anoxic interstitial waters                                                            |
| Damming                       | Enhancement of eutrophication (bottom anoxia, high organic matter in surface waters, etc.)  
Complete storage of sediments resulting in potential fish kills during sluice gate operation (high ammonia, BOD and TSS) |
| Dredging                      | Continuous high levels of TSS, and resultant silting of gravel spawning areas in downstream reaches  
Regressive erosion upstream of dredging areas may prevent fish migration |
| Felling of riparian woodland  | Continuous high levels of TSS, and resultant silting of gravel spawning areas in downstream reaches  
Increased nitrate input from contaminated groundwaters |
| Flood plain reclamation and river bed channelisation | Loss of ecological diversity, including specific spawning areas  
Loss of biological habitats, especially for fish |

BOD Biochemical oxygen demand  
TSS Total suspended solids

6.5.10 Changes in river hydrology

Many human activities, directly or indirectly, lead to modification of the river and its valley which produce changes in the aquatic environment without major changes in the chemical characteristics of the river water (Table 6.9). Such changes can lead to loss of biological diversity and, therefore, biological monitoring techniques are most appropriate in these situations, supported by careful mapping of the changes in the river bed and banks.

Major modifications to river systems include changes to depth and width for navigation, flood control ponds, reservoirs for drinking water supply, damming for hydroelectric power generation, diversion for irrigation, and canalisation to prevent loss of flood plains of agricultural importance due to river meandering. All of these affect the hydrology and related uses of the river system. The dramatic changes in the Nile after construction of the High Aswan dam are examples of both predicted and unpredicted downstream effects (Meybeck et al., 1989). Further discussion of the impoundment of rivers and the implications for assessment are given in Chapter 8.

6.6. Strategies for water quality assessment in river systems

Once polluting substances are introduced into a river, they are transported and transformed by physical, chemical, biological and biochemical processes. It is important to understand these various pathways in order to achieve the best sampling design and to determine the impact of the substance on the water system and the rates at which elimination may occur.
Sampling and analytical strategies for river assessments must be related to the present and future water uses. Two major concepts must be recognised in the design of assessment programmes which address water uses:

- Multiple use of river water may occur within any region of the river basin. Each use has different water quality requirements and user conflicts may occur. Ideally, water quality should meet the most stringent use requirements which, in virtually all cases, is the provision of good quality drinking water.

- There is always a responsibility for upstream users to ensure adequate water quality for the needs of downstream users.

Multiple use of a river system necessitates careful design of assessment programmes to ensure that the requirements of individual uses are accommodated in the monitoring strategy. Table 6.10 provides a structure for designing monitoring programmes to fit multiple water uses. Further details of the selection of specific variables are given in Tables 3.7 to 3.10. Many agencies may not be able to carry out all of the components listed in Table 6.10. Nevertheless, all the components for which facilities are available should be included and efforts should be made to upgrade monitoring capabilities until the total requirements of the scheme can be achieved routinely.

6.6.1 Physical transport and the use of bottom or suspended sediments

Physical transport is produced by riverine flow. Some pollutants may be carried in solution, in which case their transportation rates will be equivalent to the velocity of the river. There is also a tendency for dissolved pollutants to be diluted downstream, particularly if additional water derived from downstream tributaries is free of the same substance. Substances with low solubility are adsorbed by particles shortly after introduction into a river. Such substances are deposited in the river bed during low flow and are transported by resuspension at higher flows. However, most polluting substances occur in both adsorbed and soluble forms, with an equilibrium occurring between the solute and particle phases. The ratio between the two phases is subject to continuous change depending on the following conditions:

- The concentration in solution: if reduced, e.g. by biological activity, it will result in desorption from particles.

- The concentration of suspended solids: higher concentrations lead to increased particulate adsorption; lower concentrations result in less particulate adsorption.

- The composition of sediment particles including particle size, surface area and surface coatings in the provision of adsorption sites.
### Table 6.10 Selection of sampling media, frequency and water quality variable groups in relation to assessments for major uses of, or impacts on, riverine systems

<table>
<thead>
<tr>
<th>Purpose of sampling</th>
<th>Media</th>
<th>Sampling frequency</th>
<th>Filtered water &lt; 0.45 µm</th>
<th>Total unfiltered sample</th>
<th>Particulate material</th>
<th>Bacteria</th>
<th>Major ions</th>
<th>Nutrients</th>
<th>Trace elements</th>
<th>Organics</th>
<th>Suspended sediment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Potable water</td>
<td>Water</td>
<td>C</td>
<td></td>
<td></td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Industrial water supply</td>
<td>Water</td>
<td>I/F</td>
<td>x</td>
<td>x</td>
<td></td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Irrigation</td>
<td>Water</td>
<td>R</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Contact recreation</td>
<td>Water</td>
<td>S,C</td>
<td>x</td>
<td>x</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fisheries</td>
<td>Water, fish</td>
<td>R</td>
<td>x</td>
<td>x</td>
<td></td>
<td></td>
<td></td>
<td></td>
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</tbody>
</table>

**ASSESSING WATER QUALITY FOR WATER USES**

- Industrial wastewater disposal
- Municipal wastewater disposal
- Agricultural
- Urban
- Trophic status

See also Chapter 3 for more detailed information on variable selection

C Ideally continuous or daily measurements (in practice frequency is usually related to the population served)
Phosphorus provides a good example of the physical and chemical transport of a pollutant. Figure 6.26 shows the theoretical concentration profiles downstream of a sewage treatment plant effluent discharging soluble phosphorus to a river. Soluble phosphorus shows a sudden increase in concentration downstream of the effluent. Particles start to adsorb soluble phosphorus rapidly to achieve an equilibrium condition. This is followed by a depositional phase where sediments high in phosphorus accumulate in the river bed. Once in the river bed, they can be subjected to resuspension and to slow desorption of phosphorus from the river bed back into solution. The particles remain in this dynamic condition until effectively removed by permanent deposition.
Physical and chemical transport have also been illustrated for mercury in the English Wabegon river system in North West Ontario, Canada by Jackson (1980). This work is summarised in Figure 6.27. Total mercury declined with sediment dilution with progressive distance from the source and was matched by a decline in methyl-mercury in the surface water due to elimination of particles. Methyl-mercury in the mud increased downstream and was paralleled by increasing concentration in the deeper waters due to release from the bottom sediment (Figure 6.27A).

6.6.2 Use of biological methods in rivers

The possible implications of anthropogenic influences on rivers have been mentioned above. These vary from activities in the terrestrial zone (such as agriculture and deforestation), to manipulation of the aquatic zone (such as canalisation, abstraction and weirs). However, the disposal of pollutants to watercourses also has an enormous impact on the biological communities. These effects may be direct, for example by toxicity or stimulation, or indirect, such as the results of changes in oxygen concentrations. A generalised scheme of different effects on biological communities downstream of a source of organic matter, has been discussed in Chapter 5 and illustrated in Figure 5.1. The number of different kinds of organisms decreases initially and then recovers. For toxic or inert pollution, population density decreases initially and recovers slowly, but following organic, non-toxic pollution it increases very rapidly after only a slight decline.

Most methods of biological assessment described in Chapter 5 can be applied to rivers. Some ecological methods were developed specifically for use in rivers, especially in relation to organic matter pollution, e.g. saprobic indices and biotic indices. However, these methods, which are based on presence or absence of indicator species, may need
thorough testing and possible modification before use in a local river system where indigenous species may vary. Methods of ecological assessment, usually based on macroinvertebrate communities, are useful for determining the general condition of the water body or for long-term monitoring of the effects of point sources, such as sewage outfalls. An example of the use of biological assessment for classification of regional water quality is given in section 6.7.3.

Each ecological method (see Chapter 5) has specific requirements with respect to sampling the river, such as the type of sampler used and the precise conditions of flow and substrate (e.g. shallow, riffle section with stony bottom). For accurate determinations of complete benthic communities it is necessary to take sub-samples from all relevant habitats in the sampling site, e.g. stones, sand, mud, macrophytes, etc. The International Standards Organization (ISO) (ISO Standard 7828) has recommended procedures for biological surveys. Failure to adhere to appropriate procedures may prevent meaningful comparisons of the results between different rivers, or sites on the same river. However, in some situations rapid assessment may be possible in the field by examining live samples. For a more accurate calculation of a biological index samples must be kept cool and transported live to the laboratory, or they can be preserved with a suitable fixative solution. To obtain statistically accurate determinations of benthic populations for use in ecological methods it may be necessary to take a large number of samples on any single sampling occasion in order to obtain a representative population estimate (Elliott, 1977). Treatment of the data obtained during biological sampling is discussed in Chapter 10. When the number of species or individuals in samples is low, careful interpretation of the results by a trained hydrobiologist may be more meaningful than statistical treatment of the data. When there have been no major changes in a river system (such as modification of the hydrological regime or a new pollution source) a single survey of the benthic communities can represent the water quality over a long time span.

As planktonic communities are not usually well developed in river systems, methods based on samples of these organisms are not particularly suitable for biological assessment, except in laboratory-based bioassays or toxicity tests where samples of river water may be used with laboratory reared organisms (see Chapter 5). Some in-situ physiological methods, e.g. measurement of oxygen production or chlorophyll fluorescence (Figure 6.18), may be useful in lowland rivers where slow flow rates enable phytoplankton communities to develop.

The macrophyte communities of some temperate rivers have been found to be useful for indicating degrees of organic pollution (Caffrey, 1987). Some macrophytes can be readily identified without removal, sampling or transport to the laboratory and, therefore, a rapid on-site assessment of water quality may be made. However, the degrees of organic pollution indicated may be quite rough and in some cases the variety of macrophyte species may be inadequate to develop suitable indices.

The organisms of principal interest to man in rivers, as with lakes, are usually the fish. In many areas they are exploited for food or recreation, and when large numbers die they are conspicuous, indicating an obvious deterioration in water quality. Many species are top predators in the food chain and are, therefore, susceptible to food chain biomagnification of toxic contaminants (see Table 5.12). As a result they may bioaccumulate high levels of toxic compounds in their body tissues. In such situations
the fish may present a health risk to man if consumed in sufficient quantity. For this reason fish are frequently used for biomonitoring the presence of contaminants in rivers, particularly in relation to industrial discharges. The study by Jackson (1980) (see Figure 6.27B) also provides an example of biological uptake of mercury, which also represents one of the major pathways followed by pollutants in river systems. In this example, methyl-mercury concentrations in pelagic fish followed surface water concentrations, whereas benthic organism concentrations followed bottom water concentrations. Biomagnification also occurred in this river system with increasing mercury concentrations up the food chain, from crayfish (crustacea) to suckers (fish) and to walleye (fish).

As fish are usually only associated with certain riverine habitats, and are very sensitive to changes in the physical, chemical and biological quality of their habitat, a survey of the species present gives a general indication of water quality. Samples of fish can be collected with nets or traps, or by electrical methods which temporarily stun the fish (see Table 5.13). Counting and identification can be carried out in the field. The sensitivity of fish species to changes in their environment also makes them useful for bioassay procedures and toxicity tests, or for use as “early warning” organisms in dynamic or continuous bioassays (Boelens, 1987) (for further details see Chapter 5).

In certain circumstances, sampling rivers for the collection of indigenous fish or invertebrates, particularly for bioaccumulation studies or bioassays, may be difficult due to the physical nature of the river or its banks. In these situations artificial substrates, caged organisms or “dynamic” tests (see Chapter 5) are particularly useful. These methods allow comparable samples to be obtained from sites with different physical and hydrological conditions.

It must be stressed that the successful use of biological methods in water quality assessment requires the specialist knowledge of a trained hydro-biologist, together with adequately trained field personnel. A trained hydro-biologist can decide on the appropriate choice of bioindicators based on an understanding of the type of water quality information required, and on an interpretation of the biological, chemical, and physical nature of the river.

### 6.6.3 Sampling frequency

As stressed earlier, the measurement of discharge is an essential component of most sampling programmes. Without this, only qualitative surveys of the general condition of rivers can be obtained. For management of river water abstraction or water quality, a detailed background of information is necessary which includes discharge characteristics combined with an appropriate sampling strategy. Background information is particularly important with respect to diffuse sources of contaminants and for sediment associated variables which show exponential increases in concentrations with increasing discharge.

Figure 6.28 gives an example of the difficulty involved in adequately sampling a small river with a groundwater base flow and large, rapid responses of discharge to periodic rain storms and winter snow melt (see also Figure 6.3). The river (the Venoge, draining into Lake Geneva) was sampled monthly at its mouth throughout 1986 and 1987. The sampling site was adjacent to a permanent hydrographic station in order to obtain a continuous discharge record for the river. In addition to the monthly samples, this station
was sampled intensively for four storm events using a centrifuge for the recovery of high flow sediment samples, and by an automatic water sampler for five storm events (see Figure 6.28). The automatic water samples were taken to aid an understanding of storm related river water quality processes.

**Figure 6.28 Daily discharge during 1987 at Ecublens-les Bois on the Venoge river. Monthly sampling intervals and the periods sampled for storm events are also indicated (After Zhang Li, 1988)**

Based only on monthly samples, the ability to ensure adequate sampling of all river stages is severely limited. This is illustrated in Figure 6.29 where the discharge frequency is shown as a curve representing the percentage of time at which specific discharges occurred throughout the two years of the records. The number of samples taken in relation to the discharge is shown on the two curves. At discharges below approximately 9 $m^3 \cdot s^{-1}$ the water samples were collected in reasonable proportion to discharge. However, in 1986, no samples were taken at discharges greater than 9 $m^3 \cdot s^{-1}$. The discharges greater than 9 $m^3 \cdot s^{-1}$ occurred up to 14 per cent of the time and included discharges reaching 32 $m^3 \cdot s^{-1}$. Thus, the sampling design totally missed significant high flow events. Therefore, high loads of sediment, and sediment related pollutants, were not represented in the data set and the importance of these variables was severely underestimated.
Fortunately, better representation was obtained in 1987, although only two samples were taken at discharges between 12 and 16 m$^3$s$^{-1}$ which were not statistically representative. In the study of the Venoge these deficiencies were overcome by deliberate sampling during storm events. Such storm event sampling should be carried out wherever possible, particularly in small rivers with basin areas less than 1,000 km$^2$, where the random possibility of sampling high flow by constant frequency sampling is much reduced. In large rivers, with basins in excess of 100,000 km$^2$, constant frequency sampling of one month or less generally provides high quality, representative data without the need to sample specific storm events. Nevertheless, effort should be made to sample extreme events when they are adequately forecast. Extreme storm events are those which occur once in every 25 or 100 years and produce major river basin modifications.

Figure 6.30 shows the different strategies of sampling frequency in relation to different discharge measurement strategies and Table 6.11 indicates the relative cost and reliability of these monitoring strategies. Strategy A (discrete sampling without discharge data) is suitable for preliminary surveys to establish the requirements for more rigorous monitoring. Strategy B (discrete simultaneous measurement of concentration and discharge) permits calculation of instantaneous flux data and provides some information of the variability of concentration in relation to discharge. Strategy C is the most usual monitoring approach; discrete, usually equal interval, sampling for concentrations is carried out together with continuous discharge measurement. This allows a reasonable estimate of flux to be made by extrapolation of the water quality data between samples and calculation using the measured river discharge (see the example of the Venoge above). With this scheme, high discharge events are under-estimated and, therefore, the flux of sediment and related variables is also under-estimated.
In strategy D, sampling is controlled by the discharge with integrated samples taken during specified discharge rates. Continuous discharge data are available and, therefore, good average flux data may be determined. This strategy may be the optimum sampling regime that can be carried out reasonably without excessive sample numbers, although automatic sampling is necessary. A similar automated system is shown in strategy E, which is the same as strategy D except that the automatic sampler takes integrated samples over a set time period chosen in relation to discharge. Good average flux data can be obtained, but short duration events may be under-represented. This kind of system, as with strategy D, can be set up for high frequency sampling during storm events as for the automated sampling from the Venoge (see Figure 6.28). The most advanced system of river sampling is illustrated in strategy F. Continuous constituent measurement is carried out in parallel with continuous discharge recordings. This allows for continuous flux calculations and a very precisely calculated total flux for each variable measured. However, such a system is totally automated and requires the installation and maintenance of analytical probes (e.g. O₂, pH, conductivity, temperature, turbidity), automatic sampling apparatus and their associated electronic equipment. At present, few variables essential for water quality management and control can be measured on a truly continuous basis. The presence of toxic substances, however, may be monitored by continuous “dynamic” toxicity tests (see Chapter 5).

**Figure 6.30** Sampling frequency required for water quality monitoring in rivers

A. Discrete sampling; no discharge data

B. Discrete sampling; discrete discharge data; discrete flux data
C. Discrete sampling; continuous discharge; extrapolated flux record.
D. Water discharge integrated sampling; average flux data

E. Time integrated sampling; continuous discharge data; calculated average fluxes
F. Continuous records of concentration and discharge; continuous calculated fluxes.
\( C \) Concentration

\( Q \) Discharge

\( \Phi \) Flux

\( t \) Time
Table 6.11 Relative costs and benefits of various monitoring strategies for chemicals in rivers (see also Figure 6.30)

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Field operational cost</td>
<td>Hand sampling only</td>
<td>Hand sampling plus discharge measurement</td>
<td>Hand sampling plus discharge recording</td>
<td>Discharge weighted automatic sampling</td>
<td>Time weighted automatic sampling</td>
<td>Continuous concentration and discharge recording</td>
</tr>
<tr>
<td>Discharge measurement²</td>
<td>x</td>
<td>x to xx³</td>
<td>x to xx</td>
<td>x</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Capital cost of apparatus</td>
<td>Discharge recorder</td>
<td>xx</td>
<td>xx</td>
<td>xx</td>
<td>xx</td>
<td>xx</td>
</tr>
<tr>
<td>Automatic sampler</td>
<td></td>
<td></td>
<td>xx</td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Concentration recorder</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>xxx</td>
<td></td>
</tr>
<tr>
<td>Reliability</td>
<td>Time variation</td>
<td>x</td>
<td>x to xx³</td>
<td>xx</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Flux determination</td>
<td>x</td>
<td>xx</td>
<td>xx⁴</td>
<td>xx⁴</td>
<td>xxx⁴</td>
<td></td>
</tr>
</tbody>
</table>

x Low cost  
xx Medium cost  
xxx High cost

Strategies A to F are referred to in the text

¹ Depending on sampling frequency

² Establishment of gauge height-water discharge rating curve

³ Linked to sampling frequency

⁴ Reliability of flux determination when field samplers and/or concentration recorders are efficient 100 per cent of the time

As illustrated in section 4.3.3, riverine fluxes of material are highly variable in time at a given sampling station. Generally, fluxes of TSS vary more than water discharge, while most major ion fluxes vary less than water discharge (due to decreasing concentrations with increasing discharge). Consequently, the optimum frequency for discrete sampling for flux determination is influenced by these relationships. The optimum sampling
frequency is that frequency above which there is no significant gain in the accuracy of the flux determination with respect to other errors involved, such as analytical error and errors arising from the non-uniformity of the river section. Figure 6.31 illustrates the variation in the number of samples per month required for flux determination of TSS and major ions with the basin area of the river being sampled. For a given river basin the range of optimum sampling frequency is affected by the basin relief and climatic influences; steep, heterogeneous and dry basins need greater sampling frequencies than lowland, homogeneous and humid basins of the same size. In very small basins, more than four samples a day may be necessary, whereas for the Amazon and Zaire rivers, for example, a single sample a month may be sufficient.

6.6.4 Spatial distribution of samples

The spatial distribution of water quality stations within a river basin must be chosen in relation to the assessment objectives. Certain objectives, such as checking compliance with water quality guidelines for potable supply or other specific uses, require samples for determination of concentrations. For protecting water quality further downstream, e.g. in lakes, estuaries or the sea, it is necessary to know the loads or fluxes of river components.

The location of sampling stations should coincide with, or at least be near to, discharge measurement stations. In addition, water quality stations should be placed immediately upstream and downstream of major confluences and water use regions (e.g. urban centres, agricultural areas including irrigation zones, impoundments and major industrial complexes). Such locations are illustrated schematically in Figure 4.1. Many rivers cross national, state or municipal boundaries and the responsible monitoring authorities should ensure that stations are also located at the boundaries of river input and river output for their particular regions. Such stations provide information on the suitability of water for use within the region, and determine whether water of inferior quality is being exported to downstream users.
A single station at the mouth of a river, at which water quality variables are measured, may be adequate for flux calculations. Such a station must be located far enough upstream of the junction with the receiving water body to prevent changes in level or discharge due to tidal or water level changes within the receiving water body. At the same time there must be no significant pollution inputs between the monitoring station and the receiving water body.

Very large rivers can be several kilometres wide at their downstream reaches and, therefore, sampling should be carried out through several vertical profiles across the river at each station. Vertical profile samples are normally mixed to give depth integrated measurements. Small and medium sized rivers may be sampled at one location provided that the river is well mixed and homogeneous with respect to water quality.
Table 6.12 Relative homogeneity in the concentration of suspended sediment and its composition (as inferred from the apatite phosphorus) across the River Rhône for three stations and three sampling episodes at Porte du Scex (samples taken at 0.2 m depth)

<table>
<thead>
<tr>
<th>Date</th>
<th>Station</th>
<th>Suspended sediment (mg l⁻¹)</th>
<th>Apatite phosphorus (µg g⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>60 Oct 1981</td>
<td>Right bank</td>
<td>58</td>
<td>460</td>
</tr>
<tr>
<td></td>
<td>Centre</td>
<td>60</td>
<td>485</td>
</tr>
<tr>
<td></td>
<td>Left bank</td>
<td>53</td>
<td>360</td>
</tr>
<tr>
<td>20 Oct 1981</td>
<td>Right bank</td>
<td>73</td>
<td>410</td>
</tr>
<tr>
<td></td>
<td>Centre</td>
<td>79</td>
<td>430</td>
</tr>
<tr>
<td></td>
<td>Left bank</td>
<td>77</td>
<td>460</td>
</tr>
<tr>
<td>4 Nov 1981</td>
<td>Right bank</td>
<td>36</td>
<td>410</td>
</tr>
<tr>
<td></td>
<td>Centre</td>
<td>28</td>
<td>355</td>
</tr>
<tr>
<td></td>
<td>Left bank</td>
<td>36</td>
<td>375</td>
</tr>
</tbody>
</table>

Source: After Burrus et al., 1989

With the exception of small rivers, the location of any sampling site needs to be evaluated for its suitability to represent that section of the river. This is normally done by means of a detailed study of a cross-section of the river. Ideally, the cross-section should be downstream of a physical structure resulting in mixing of the water, e.g. rapids, waterfalls or river bends producing a helical mixing pattern. The cross-section has to be checked for the efficiency of the mixing process by analysing a minimum of three locations within the cross-section: the left and right banks and the centre of the river. At each location a minimum of three depth samples should be taken: one below the surface, one in mid-column and one above the river bed (above the traction or carpet load). If these samples are homogeneous, then the single sampling site can be located at any point in the cross-section and can be assumed to be representative of the river water quality. An example of site evaluation has been described for the mouth of the Upper Rhône river on Lake Geneva by Burrus et al. (1989). By measuring particulate apatite phosphorus in sediments they were able to show considerable homogeneity of sediment composition throughout the cross-section of the river (Table 6.12).

Laminar flow may confine a water mass to one river bank downstream of a major effluent (see Figure 6.7). When sampling is undertaken specifically to monitor the effects of the effluent, the sample site can be located downstream, on the same side of the river as the effluent. Sites further downstream, after the point of mixing or overturn, need to be chosen as described above. In large rivers a well-mixed site may be several kilometres below the source of the effluent.

6.6.5 Sampling methods

Most methods of river sampling are based on a bottle collection or water pump system. Difficulty can be experienced in collecting water samples from very wide rivers. It may be
necessary to use a boat if samples cannot be taken from a bridge or to use a specially constructed cable system. When homogeneity of water quality has been established for the cross-section of the river, it may be possible to take certain samples from the river bank at fairly deep locations. However, the velocity of the water may make it difficult to submerge a water sampling device adequately. This may be overcome by using hydrodynamically designed samplers (streamlined and fish-like in shape) which have a depressor fin and tail to help maintain the correct orientation in relation to the flow of the water.

Details of field techniques for collecting samples are described in Bartram and Ballance (1996) and many kinds of water sampling devices are described in operational guides (e.g. WHO, 1992) or manufacturers’ catalogues. Most devices operate by filling the sample container at a fixed depth or allowing operation such that a depth integrated sample can be obtained. Water pump intakes can also be set at fixed depths. Alternatively, a depth integrated sample can be obtained by hauling the pump inlet through fixed depths and allowing pumping to continue for a fixed time interval at each depth. Pumps can be used in this way to collect a series of depth integrated samples across a river, thereby giving a fully integrated, large volume sample for the river cross-section.

There are many automated systems available for sampling and analysing river water. The instrument used for the study of the Venoge mentioned above (section 6.6.3), could be powered externally or by internal batteries. It contained 24 one-litre bottles which were filled by a pumping system which could be set to operate at selected time intervals. This system allows the collection of 24 individual samples, or for each bottle to be filled for several small time intervals to provide 24 integrated one-litre samples. Alternatively, sampling may be triggered by a water level sensing device and samples may be collected for pre-selected time intervals during high discharge events. As such automated sampling instruments may be small, portable and versatile, they may be used for synoptic sampling in river basins or to monitor the changes in a flood wave as it progresses through the watershed. Similar systems with more numerous water bottles can also be obtained for permanent installation in association with discharge measurement stations.

Truly continuous water quality monitoring can only be carried out with submersible probes. Although these are only available for selected variables, e.g. O₂, pH, temperature, conductivity and turbidity, they provide very useful information. For security purposes the probes may be immersed in a flow of water pumped through a by-pass system housed within the gauging station. Alternatively, the probes may be suspended in the river from an anchored flotation system or overhead structure. The data can be continuously recorded using a pen and paper plotter or microcomputer. Highly sophisticated systems allow the digital data to be sent electronically to a central recording location or via satellite transmission to a distant location.

Sediment sampling is usually carried out by filtering water samples collected by the methods mentioned above. Filtration is normally done with 0.45 µm pore diameter filters, but these are subject to pore clogging. Such samples are suitable for determining total suspended solids but the final sample size is usually too small for comprehensive chemical analyses, e.g. of several trace elements. To overcome this problem, continuous flow centrifuges have been adapted for field use. Such instruments can
process 6 litres per minute of water, with a recovery efficiency of approximately 98 per cent of particles greater than 0.45 µm and 90-95 per cent of particles greater than 0.25 µm. Many tens of grams of material can be obtained, even at low suspended sediment concentrations, by processing very large volumes of sample. Santiago et al. (1990) were able to obtain sufficient material for analysis of particle size, organic and inorganic carbon, nitrogen, forms of phosphorus, trace elements and organic pollutants, even from samples with a suspended solids concentration of 6 mg l⁻¹.

6.6.6 Laboratory techniques

Analytical techniques for water and sediment are described in Bartram and Ballance (1996) and various other specialised manuals. However, some general information is available in Chapters 3 and 4, including guidance on sample preservation methods.

An example of a river assessment programme from Germany, indicating the stages undertaken within the field and laboratory, is given in Figure 6.32. This system is based on comprehensive chemical analysis of water samples and excludes suspended sediment analysis. It is advisable to construct a flow diagram similar to Figure 6.32 for any monitoring programme prior to commencing the field work. This will ensure that an adequate quantity of sample is collected for all subsequent analyses.
In the development of environmental quality assessment networks, multipurpose monitoring is often the first type to be established. However, this may eventually be developed into more specialised or sophisticated activities, such as basic surveys, trend monitoring or operational surveillance. An example of such a development is given in Table 6.13 from the Nordrheine-Westfalen region in Germany. However, it is more common for each monitoring programme to be designed by a different group of experts, each involved in different aspects of water quality management. Occasionally, the field and laboratory specialists are not involved in the preparation and planning of monitoring activities. Consequently, similar sampling and analysis can frequently be undertaken by two or more agencies with interests in water quality (e.g. agencies for agriculture, housing and development, fisheries and natural resources, environment or health) and sometimes within the same agency. In addition, monitoring may be repeated by non-governmental organisations, such as private water treatment and supply companies. It is, therefore, highly recommended that:
• all information on water quality assessment is centralised by a coordinating authority for each main river basin,

• basic data should generally be freely accessible from programmes financed by public means,

• common data banks should be established for river basins, at either national or inter-state level, and

• all monitoring activities having similar objectives should be co-ordinated and harmonised to ensure maximum inter-comparability and proper interpretation (e.g. eutrophication monitoring and assessment, major trend stations, micropollutant monitoring in sediments).

**Table 6.13 Example of a water quality monitoring system: Nordrhine-Westfalen, Germany**

<table>
<thead>
<tr>
<th>Operations</th>
<th>Frequency</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nordrhine-Westfalen</td>
<td>Area: 34,000 km²</td>
</tr>
<tr>
<td></td>
<td>Total length of rivers: 60,000 km</td>
</tr>
<tr>
<td></td>
<td>Population density: 500 km²</td>
</tr>
<tr>
<td></td>
<td>Rhine river stretch: 200 km (from km 639 to km 863)</td>
</tr>
<tr>
<td><strong>BASIC MONITORING</strong></td>
<td></td>
</tr>
<tr>
<td>Number of points</td>
<td>Approximately 3,000</td>
</tr>
<tr>
<td>Programme (frequency)</td>
<td>Analysis of benthic organisms</td>
</tr>
<tr>
<td></td>
<td>Calculation of saprobic index</td>
</tr>
<tr>
<td></td>
<td>Analysis of basic physical-chemical variables: conductivity, temperature, pH, BOD, NH₄, Cl</td>
</tr>
<tr>
<td>Data treatment</td>
<td>Classification into a seven stage water quality system. Water-quality report</td>
</tr>
<tr>
<td></td>
<td>Water-quality map</td>
</tr>
<tr>
<td><strong>IMPACT MONITORING</strong></td>
<td></td>
</tr>
<tr>
<td>Number of points</td>
<td>Approximately 150 in selected catchment areas</td>
</tr>
<tr>
<td>Programme (frequency)</td>
<td>Analysis of benthic organisms</td>
</tr>
<tr>
<td></td>
<td>Intensive chemical analysis: variables adapted to the special problems of the water course: basic variables, trace elements, organic micropollutants, etc.</td>
</tr>
<tr>
<td>Data treatment</td>
<td>Report as basis for management and control measures</td>
</tr>
<tr>
<td><strong>TREND MONITORING</strong></td>
<td></td>
</tr>
<tr>
<td>Number of points</td>
<td>Approximately 50 at selected sites</td>
</tr>
<tr>
<td>Programme (frequency)</td>
<td>Analysis of benthic organisms</td>
</tr>
<tr>
<td></td>
<td>Intensive chemical analysis</td>
</tr>
</tbody>
</table>
6.7. Approaches to river monitoring and assessment: case studies

River assessments have increased greatly in number and sophistication over the last 30 years. The traditional practice of multi-purpose monitoring (see Chapter 2) is not always adequate for specific water quality issues. Many recent water quality assessment activities have combined the analyses of water, sediments, and biota.

6.7.1 Preliminary surveys

Before establishing the appropriate surveys of trace elements in the River Seine, France a very detailed survey was carried out at the river mouth station situated at Poses. All the necessary precautions were taken to avoid contamination during sampling, pretreatment and analysis (D. Cossa and M. Ficht, 1991 pers. comm.). As a result, the dissolved Cd concentrations were found to be more than ten times lower than previously reported in routine analyses. The variability was established on a weekly basis, pointing out a relative dilution during the high water stage and an unexpected peak in the summer (Figure 6.33). Consequently, weekly sampling was eventually chosen as appropriate for dissolved Cd since concentrations were much lower than water quality criteria and about 90 per cent of the total Cd discharged by the Seine was associated with particulate matter (D. Cossa, pers. comm.). Future monitoring should probably
focus on suspended matter which can be analysed more easily and less frequently (e.g. bimonthly).

6.7.2 Multi-purpose monitoring

Multi-purpose monitoring should provide answers to many diverse questions concerning water quality at a fixed point on a river, or concerning the river water quality of a given region. Typical questions are:

- Is the water fit for drinking?
- Is the water fit for other major uses?
- Is the quality of the aquatic environment adequate to support the growth of the expected aquatic biota?
- What are the time variations in water quality?
- What are the long-term trends in water quality?
- What is the flux of pollutants (or nutrients) in the river?
- Where are the major pollutant sources (diffuse and point sources)?
- What is the regional distribution of water quality?
- Are pollution control measures adequate?

Each of these questions actually requires a specific monitoring activity and, therefore, multi-purpose monitoring is usually a compromise based on average station densities, average sampling frequencies and a restricted number of variables (depending on the financial resources of the monitoring agency). If the station density is mostly controlled by the technical and financial means available, the frequency and number of variables should be chosen by scientists in relation to the assessment objectives. Table 6.14 proposes a basis for the development of a multi-purpose river monitoring scheme in relation to the three stages of monitoring complexity discussed in Chapter 2.

6.7.3 Basic surveys

Basic surveys are used mainly when detailed spatial distribution of water quality is needed, perhaps as a starting point against which future changes can be measured. The accent is put on the station density rather than on sampling frequency. This enables the drawing of water quality maps or detailed river profiles identifying the various changes in quality. This type of water quality assessment requires a preliminary survey (inventory) of all potential sources of pollution in order to locate monitoring stations appropriately. Typical basic surveys combine all three media: biota, water and sediments.

The inter-comparability between stations for a given survey, and between surveys from one year to another, should always be carefully checked. As a result these surveys
should not be carried out at periods when the aquatic system is very variable, such as during high water discharge, or at the beginning of biological cycles (Hines et al., 1976). In the northern hemisphere, the end of the summer period is a suitable sampling time provided that there are no rain storms. The density of sampling stations in a given river basin can often be ten times higher than the number of regular or multi-purpose monitoring stations, but they may be sampled only once a year, or even less if no major environmental change is expected.

**Table 6.14** Development of a multi-purpose river quality monitoring programme for three different levels of monitoring complexity

<table>
<thead>
<tr>
<th>Number of stations</th>
<th>Sampling frequency (per annum)</th>
<th>Water analysis</th>
<th>Sediment analysis</th>
<th>Biological surveys</th>
<th>Required resources</th>
</tr>
</thead>
<tbody>
<tr>
<td>SIMPLE MONITORING</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>10</td>
<td>6</td>
<td>°C, pH, Cond., O₂, TSS, major ions, NO₃, visual observation</td>
<td></td>
<td></td>
<td>Small sampling team, general analytical chemistry laboratory¹</td>
</tr>
<tr>
<td>INTERMEDIATE MONITORING</td>
<td>100</td>
<td>6 to 12</td>
<td>as above, plus PO₄, NH₄⁺, NO₂⁻, BOD, COD</td>
<td>Trace elements</td>
<td>Biological indices</td>
</tr>
<tr>
<td>ADVANCED MONITORING</td>
<td>100 to 1,000</td>
<td>&gt; 12</td>
<td>as above, plus soluble organic pollutants, DOC, POC, chlorophyll, some trace elements</td>
<td>as above, plus organic micropollutants</td>
<td>as above, plus chemical analysis of target organisms</td>
</tr>
</tbody>
</table>

Intermediate and advanced monitoring needs to be combined with hydrological surveys, especially continuous records of water discharge

TSS Total suspended solids  
BOD Biochemical oxygen demand  
COD Chemical oxygen demand  
DOC Dissolved organic carbon  
POC Particulate organic carbon

¹ As required for agriculture or health departments

Water analyses for basic surveys usually include all general variables, i.e. nutrients, BOD, COD and major ions (see Chapter 3). The occurrence of micropollutants can be checked using deposited sediments from the river bed. A detailed visual inspection should also be made and biological samples collected (usually of benthic organisms) for eventual identification and counting in the laboratory. The results can be used to determine appropriate biological indices (see Chapter 5). In order to achieve maximum
inter-comparability of results from these surveys, it is highly recommended that a limited number of field and laboratory personnel be used.

The final output of basic surveys is often presented in the form of water quality maps based on chemical data or biological assessment methods or longitudinal profiles of the river course. Figures 6.34 and 6.35 show the improvements, using colour coding, in water quality over 20 years in Nordrhine-Westfalen, Germany based on the biological classification described in Table 6.15. During these 20 years measures to improve water quality included the installation and maintenance of sewage collection and treatment facilities.

### 6.7.4 Operational surveillance

Operational surveillance provides local and regional planning and environmental management authorities with data for such purposes as checking compliance with guidelines and legislated standards and the efficiency of water and environmental protection measures. Such monitoring may be characterised by a high station density in a restricted area. For checking compliance with guidelines for a particular water use, continuous records of pH, oxygen and other variables (e.g. ammonia) amenable to automatic sampling are frequently used at the point of water intake. In some cases the same results can be used to set-up or validate water quality models of pollutant dispersion, oxygen balance, etc.

### 6.7.5 Trend and flux monitoring

Trend and flux monitoring are both characterised by a high sampling frequency. Trend monitoring may consider the time-averaged concentrations over the whole year, or the average concentrations at special sampling periods only, such as at low water when the dilution of pollutants from a point source is minimum, during summer periods when eutrophication effects are greatest, or during floods to check the maximum levels of total suspended solids and associated pollutants. For determination of trends (see also Chapter 10) it is important that samples are collected at (or data averaged for) equally spaced time intervals (e.g. annually, monthly). Before such monitoring programmes begin, preliminary surveys are conducted to determine the time-variability of the concentrations of interest in order to help optimise sampling regimes. The sampling frequency is much higher for compounds with highly variable concentrations, such as NH₄⁺, than for more stable ones, such as Ca²⁺. Where appropriate sampling frequencies cannot be attained, the annual average concentration may not be very precise and may prevent the establishment of trends, particularly if the rate of change over the years is lower than the precision of the values. For example, a 10 per cent annual increase in ammonia will not be apparent for many years if the precision of the annual average ammonia concentration is ± 30 per cent. An example of a long-term trend is given in Figure 6.22.
Table 6.15 Definition of quality grades used to classify running waters in Nordrhine-Westfalen, Germany

<table>
<thead>
<tr>
<th>Quality grade I:</th>
<th>unpolluted to very mildly degraded¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sections of water bodies with pure, almost always oxygen saturated and nutrient-poor water; low bacteria content; moderately densely colonised, mainly by algae, mosses, flatworms and insect larvae; spawning ground for salmonid fish.</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Quality grade I-II:</th>
<th>mildly degraded</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sections of water bodies with slight inputs of organic or inorganic nutrients but without oxygen depletion; densely colonised mostly by diverse species but dominated by salmonid fish.</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Quality grade II:</th>
<th>moderately degraded¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sections of water bodies with moderate pollution but a good oxygen supply; a very great variety and density of individual species of algae, snails, crayfish and insect larvae; considerable stands of macrophytic plants; high yields of fish.</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Quality grade II-III:</th>
<th>critically degraded¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sections of water bodies with inputs of organic, oxygen consuming substances capable of producing critical oxygen depletion; fish kills possible during short periods of oxygen deficiency; declining numbers of macro-organisms; certain species tend to produce massive populations; algae frequently cover large areas; usually high-yields of fish.</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Quality grade III:</th>
<th>heavily polluted</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sections of water bodies with heavy organic, oxygen depleting pollution; usually low oxygen content; localised deposits of anoxic sediment; filamentous sewage bacteria and colonies of non-motile ciliated protozoa predominate over the growths of algae and higher plants; occasional mass development of a few micro-organisms which are not sensitive to oxygen deficiency such as sponges, leeches and water lice; low fish yields; periodic fish kills occur.</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Quality grade III-IV:</th>
<th>very heavily polluted</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sections of water bodies with substantially restricted living conditions resulting from very severe pollution with organic, oxygen depleting substances, often combined with toxic effects; occasional total oxygen depletion; turbidity from suspended sewage; extensive anoxic sediment deposits, densely colonised by red blood-worm larvae or sediment, tube-dwelling worms; a decline in filamentous sewage bacteria; fish generally not present, unless only locally.</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Quality grade IV:</th>
<th>excessively polluted</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sections of water bodies with excessive pollution by organic, oxygen depleting sewage; processes of putrefaction predominate; prolonged periods of very low oxygen concentrations or total deoxygenation; mainly colonised by bacteria, flagellates and mobile ciliates; no fish stocks; loss of biological life in the presence of severe toxic inputs.</td>
<td></td>
</tr>
</tbody>
</table>

¹ Degraded refers to a slight deterioration in water quality which does not usually affect normal aquatic life and is considered acceptable, whereas polluted water causes the loss of most aquatic species and is unacceptable (see also section 1.2).
For flux monitoring, the sampling effort should be concentrated on the high flow periods because the maximum transport of any dissolved or particulate compounds occurs during maximum water discharge (see also section 6.6.3). Even for a constant source of pollutants, the dilution process is never perfect. Preliminary surveys should be used to determine the period of time responsible for any given percentage of the flux. A complete theoretical example is treated in Table 6.16. Flux monitoring for element A (which could, for example, be the total suspended matter and associated pollutants) should be focused on the 42 days of high discharge which correspond to 76 per cent of the annual flux. However, monitoring for element B (characteristic of dilution of a point-source pollutant in a river) should be spread over the whole year although, due to the imperfect dilution characteristics, 53 per cent of the river flux occurs during 31 per cent of the time. In both cases, the optimal number of samples should be taken in proportion to the flux distribution. For example, when taking 50 samples a year, in the case of A, 27 of them should be taken during the 12 days of maximum river water discharge (these can correspond to one, or several major floods). This may present some operational problems. However, automatic sampling can help simplify field work, although the equipment requires regular maintenance, usually on a weekly basis (see section 6.6.5).

Table 6.16 Flux monitoring: hypothetical example of discharge and flux distribution for elements increasing (A) and decreasing (B) with water discharge (Q)

<table>
<thead>
<tr>
<th>Gauge height duration</th>
<th>Water discharge</th>
<th>Concentration (by vol)</th>
<th>% of annual flux</th>
</tr>
</thead>
<tbody>
<tr>
<td>Days</td>
<td>% of year</td>
<td>Average Q(^1)</td>
<td>% annual volume</td>
</tr>
<tr>
<td>2</td>
<td>0.6</td>
<td>1,000</td>
<td>5.6</td>
</tr>
<tr>
<td>10</td>
<td>2.7</td>
<td>500</td>
<td>13.9</td>
</tr>
<tr>
<td>30</td>
<td>8.3</td>
<td>250</td>
<td>20.9</td>
</tr>
<tr>
<td>70</td>
<td>19.2</td>
<td>125</td>
<td>24.4</td>
</tr>
<tr>
<td>253</td>
<td>69.3</td>
<td>50</td>
<td>35.2</td>
</tr>
<tr>
<td>Totals</td>
<td>365</td>
<td>100.0</td>
<td>100.0</td>
</tr>
</tbody>
</table>

\(^1\) Arbitrary units

For trend monitoring, individual samples can be combined to form time-averaged composite samples or the individual measurements can be combined to give a mean value. For flux monitoring, composite samples must be weighted according to the water discharge at the time of collection of each individual sample.

Long-term trends in rivers can also be assessed by the study of sediments from flood plains or old river channels. Figure 6.23 shows the changes in pollutant concentrations in a sediment core taken near the mouth of the old Rhine river in 1978 and dated with a \(^{137}\)Cs profile. The most marked increase in contaminants (organic micropollutants) within the core occurred after 1940.

6.7.6 Early warning surveillance and associated networks

Continuous early warning surveillance is mostly carried out at water intakes for major drinking water treatment plants. It must give instantaneous (or very rapidly obtained)
information on the overall water quality, especially the presence of toxic substances. Continuous measurement of dissolved substances is highly expensive and usually sophisticated. Other approaches are based on the continuous exposure of very sensitive organisms to river water. These biological monitoring methods are often based on the behaviour or activity of organisms such as fish (e.g. trout) or large invertebrates such as mussels and water fleas (e.g. *Daphnia* spp.) (see section 5.7.3). A biological early warning system is being developed for the Rhine using a suitable biotest for each trophic level (Schmitz *et al*., 1994).

Automatic, continuous, chemical analysis is relatively cheap and easy for certain variables, such as dissolved oxygen, temperature, electrical conductivity, optical turbidity and pH, but is more difficult for ammonia, dissolved organic carbon and some specific major ions. It is very expensive and complicated for trace elements and organic pollutants, in addition to which the detection limits are relatively high. These factors restrict the use of automatic, continuous analysis for trace elements and organic pollutants to early warning stations.

An early warning network may be established on large rivers linking key stations upstream and downstream of water intakes. This is particularly useful for river stretches affected by major industrial areas, nuclear power plants, etc. Such networks are activated in the event of accidents (e.g. chemical spills and burst storage tanks), decreased efficiency of effluent treatment plants or unusual meteorological events (such as thunderstorms causing increased urban run-off with high BOD and resultant oxygen depletion in the river). Early warning networks are useful, not only for protecting major drinking water intakes, but also for protecting commercial fisheries, irrigated fields and livestock watering sources.

An example of an early warning system exists in Germany for the River Rhine which is an international water body characterised by extensive industrial and urban development. In addition, the river is an international shipping waterway and supplies 20 million people with drinking water. Therefore, a well organised, international warning system has been installed to help prevent accidental pollution reaching critical water intakes, and it has now been working successfully for many years. The system consists of eight main warning centres. The warning message may follow either a step by step route along the river course from one warning centre to the next, or a general alert may be given to all downstream stations. Each warning message is first passed on by telephone and then in written form by telex, telefax or letter. The message includes the following information:

1. Name of person, agency, etc., giving the message.
2. Date and time of accident.
3. Place of accident (i.e. name of place, kilometre position along the river, location (right, left or centre) and any other details).
4. Type of substance, the chemical compound involved, etc.
5. Quantity of substance discharged to the river, extent of accident and pollution effects.
6. Type of pollution effect (e.g. fishkill, colour, smell).
7. Preliminary actions taken.

After collecting the initial information more detailed data have to be provided by specialists in order to assess the impact and plan remedial action. This information includes:

- Quality criteria of the substance in question (e.g. solubility, density and reaction with water, air or other compounds).

- Potential effects on water quality, toxicity to man, fish or the ecosystem, self-purification ability and the risk category in water.

- Hydrological conditions in the river (discharge in m$^3$ s$^{-1}$ and current velocity in m s$^{-1}$).

- Measured and calculated concentrations at the place of the accident, and downstream at any important sites. The expected time of arrival of the contaminant at given places such as drinking water intakes. Any initial action taken to minimise the effects.

When an alarm has been given but the danger has passed, it is also necessary to give an all-clear signal. Finally, it must be emphasised that in order to be effective early warning surveillance should be backed up by technical provisions (such as equipment for aeration) and by legal powers to initiate and carry out necessary preventative action (such as stopping waste disposal activities).

6.7.7 Surveys for water quality modelling

Water quality modelling can be a valuable tool for water management since it can simulate the potential response of the aquatic system to such changes as the addition of organic pollution, the building of small hydro-electric power plants, the increase in nutrient levels or water abstraction rates and changes in sewage treatment operations (such as the addition of tertiary treatment). For major projects the cost, including the necessary surveys, of producing a model can be only a few per cent of the total cost of the new operations or activities to be introduced into the river basin. The use of generally available models (i.e. models not produced as part of a specific project) should be carried out with caution and, where possible, any general model should be verified with data obtained from the water body for which its use is being considered.

Most existing river models are for oxygen balance and are based on BOD measurements, although more recent models include the influence of phytoplankton, macrophyte growth and benthic respiration. Models based on bacterial respiration are now also being developed (Billen, 1990 pers. comm.). Obtaining the necessary data for construction or verification of the models may require additional surveys, together with data from operational surveillance and multi-purpose monitoring networks. It is important to verify models if they are to be used routinely in the management of water quality (e.g. Rickert, 1984).

Krenkel and Novotny (1980) have listed the categories of variables required for oxygen balance modelling as:
• hydrological variables (e.g. river discharge),
• hydraulic variables (flow velocity, geometry of river bed, turbulence, etc.),
• oxygen sinks (e.g. benthic oxygen demand, nitrification of ammonia),
• oxygen sources (e.g. re-aeration, atmospheric exchange, primary production), and
• temperature.
Downstream of sewage effluent discharges from treatment plants using biological, secondary processes, bacterial activity may also need to be incorporated into the models.

6.8. Summary and conclusions

Rivers have to support a wide variety of activities including water supply for various uses (drinking water and irrigation of agricultural land being amongst the most important). Progressive urbanisation and industrial development has also led to increasing use of rivers for waste disposal activities. The pollution arising from these and other sources, such as use of agricultural pesticides, has led to the increasing need for rigorous assessment of river water quality. The complexity and components of such assessment programmes are defined by the water uses and their water quality requirements, as well as a need to protect the aquatic environment from further degradation.

Rivers are dynamic systems which respond to the physical characteristics of the watershed, which in turn are controlled by the local and regional geological and climatic conditions. The size of the watershed controls the fluctuations in water level, velocity and discharge. Extreme or rapid fluctuations are dampened as watershed size increases. The flow characteristics of a river are important to the understanding of the water mixing processes in the river channel, i.e. in association with laminar flow, helical overturn and turbulent mixing in channels with rapids and waterfalls. A basic knowledge of these processes is necessary for the correct siting of sampling stations within the watershed.

Determination of river discharge is extremely important for the measurement of the flux of material carried by the river and transported to downstream receiving waters. The changing concentrations of chemical variables relative to the changing volume and velocity of river waters can provide useful diagnostic information on the origins of contaminants. In general, with increasing discharge point sources of contaminants are diluted whereas diffuse sources show increased concentrations. Sediment related variables fluctuate with suspended solids concentrations which in turn are related to discharge.

Rivers can be characterised by particular communities of organisms which are dependent on certain conditions of discharge and the physical, chemical and structural effects that it has on the river bed and water quality. Changes in the structure or quality of the river resulting from anthropogenic activities often produce specific changes in the biological communities which, once identified, can be used to monitor changes in the river environment.

The media (water, sediment or biota) which are selected for monitoring depend on the environmental properties of the chemical variables of interest and the objectives of the programme. Sediment associated variables such as phosphorus, trace elements and some organic pollutants may be best evaluated by collection of suspended sediment
samples using filtration or centrifugation methods. Contaminants occurring at very low dissolved concentrations, but which are accumulated in biota, may be studied by the collection and analysis of biological material, particularly if they are also accumulated by food organisms for potential human consumption.

6.9. References


Billen, G. 1990 GMMA, Université Libre de Bruxelles. Personal communication.


FEEMA 1987 Qualidade das Aguas do Estado do Rio de Janiero, Brazil. Fundaçao de Tecnologia de Saneamento Ambiental, Rio de Janiero.


Figure 6.34 Biological assessment and classification (see Table 6.15) of river water quality in the State of Nordrhein-Westfalen, Germany during 1969/70. Compare with Figure 6.35.
Figure 6.35 Biological assessment and classification (see Table 6.15) of river water quality in the State of Nordrhein-Westfalen, Germany during 1989/90. Improvements in water quality since 1969/70 (Figure 6.34) are due to effective water quality control over the 20 year period.