discharges directly to surface waters, whereas *user charges* are levied for the use of the authority's collective treatment system (Table 1.16). By charging industry for treating their effluents in terms of strength and volume, it encourages them to optimise production efficiency by reducing the volume and strength of their effluent. Most important of all, such charging systems ensure that effluent disposal and treatment costs are taken into account by manufacturers in the overall production costs, so that the cost of the final product reflects the true cost of production (Deering and Gray 1986).

Wastewater treatment is not solely a physical phenomena controlled by engineers, it also involves a complex series of biochemical reactions involving a wide range of micro-organisms. The same micro-organisms that occur naturally in rivers and streams are utilised, under controlled conditions, to rapidly oxidise the organic matter in wastewater to innocuous end products that can be safely discharged to surface waters. Compared with other industries which also use micro-organisms, such as brewing or baking, wastewater treatment is by far the largest industrial use of micro-organisms using specially constructed reactors. As treatment plants that were constructed during the early expansion of wastewater treatment in the late nineteenth and early twentieth centuries now near the end of their useful lives, it is clear that the opportunities for the biotechnologists to apply new technologies, such as genetic manipulation combined with new reactor designs, to pollution control are enormous (Chap. 10). In the future, cheaper, more efficient, and more compact processes will be developed, with the traditional aims of removing organic matter and pathogens to prevent water pollution and protect public health replaced with a philosophy of environmental protection linked with conservation of resources and by-product recovery (Chap. 11).

Natural scientists, whether they are trained as microbiologists, biochemists, biologists, biotechnologists, environmental scientists or any other allied discipline, have an important role in all aspects of public health engineering. They already have a significant function in the operation and monitoring of treatment plants, but their expertise is also needed in the optimisation of existing plants and in the design of the next generation of wastewater treatment systems.

## 1.2. Nature of Wastewater

Although there has been a steady increase in the discharge of toxic inorganic and organic materials, it is still the biodegradable organic wastes that are the major cause of pollution of receiving waters in Britain and Ireland (Gray and Hunter 1985; DETR 1998; Environment Agency 1998, 1999; EPA 2000). Organic waste originates from domestic and commercial premises as sewage, from urban runoff, various industrial processes and agricultural wastes. Not all industrial wastes have a high organic content that is amenable to biological treatment, and those with a low organic content, insufficient nutrients, and which contain toxic compounds, require specific chemical treatment, such as neutralisation, chemical precipitation, chemical coagulation, reverse osmosis, ion-exchange, or adsorption onto activated carbon (Table 1.8) (Casey 1997).

This book concentrates on non-toxic wastewaters. It is these that are of particular interest to the biologist and biotechnologist in terms of reuse, conversion, and recovery of useful constituent materials. Primarily sewage containing pathogenic micro-organisms is considered, although other wastewaters, such as agricultural wastes from intensive animal rearing and silage production, food processing wastes, and dairy industry wastes are also briefly reviewed.

### 1.2.1. *Sources and variation in sewage flow*

The absolute minimum quantity of wastewater produced per person (per capita), without any excess water, is 4 litres per day. At this concentration, the wastewater has a dry solids content in excess of 10%. However, in most communities that have an adequate water supply this minimum quantity is greatly increased. In those countries where technology and an almost unlimited water supply has led to the widescale adoption of water-consuming devices — many of which are now considered to be standard, if not basic, human requirements — the volume of wastewater produced has increased by a factor of 100 or more. Flush toilets, baths, showers, automatic washing machines, dishwashers and waste disposal units all produce vast quantities of diluted dirty (grey) water with a very low solids content and all requiring treatment before being discharged to surface waters. For example, a flush toilet dilutes small volumes of waste matter ($<$ 1 litre) to between 10 or 30 litres each time it is used. Domestic sewage is diluted so much that it is essentially 99.9% water with a dry solids content of less than 0.1%. Conventional sewage treatment aims to convert the solids into a manageable sludge (2% dry solids) while leaving only a small proportion in the final effluent (0.003% dry solids).

The total volume of wastewater produced per capita depends on the water usage, the type of sewerage system used and the level of infiltration.

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Table 1.8. Main chemical and biological unit processes employed in wastewater treatment.

| Process | Description |
|---|---|
| *Chemical unit processes* | |
| Neutralisation | Non-neutral waste waters are mixed either with an alkali (e.g. NaOH) or an acid (e.g. $H_2SO_4$) to bring the pH as close to neutral as possible to protect treatment processes. Widely used in chemical, pharmaceutical and tanning industries |
| Precipitation | Dissolved inorganic components can be removed by adding an acid or alkali, or by changing the temperature, by precipitation as a solid. The precipitate can be removed by sedimentation, flotation or any other solids removal process |
| Ion-exchange | Removal of dissolved inorganic ions by exchange with another ion attached to a resin column. For example Ca and Mg ions can replace Na ions in a resin, thereby reducing the hardness of the water |
| Oxidation reduction | Inorganic and organic materials in industrial process waters can be made less toxic or less volatile by subtracting or adding electrons between reactant (e.g. aromatic hydrocarbons, cyanides, etc.) |
| *Biological unit processes* | |
| Activated sludge | Liquid waste water is aerated to allow micro-organisms to utilise organic polluting matter (95% reduction). The microbial biomass and treated effluent are separated by sedimentation with a portion of the biomass (sludge) returned to the aeration tank to seed the incoming waste water |
| Biological filtration | Waste water is distributed over a bed of inert medium on which micro-organisms develop and utilise the organic matter present. Aeration occurs through natural ventilation and the solids are not returned to the filter |
| Stabilisation ponds | Large lagoons where waste water is stored for long periods to allow a wide range of micro-organisms to break down organic matter. Many different types and designs of ponds including aerated, non-aerated and anaerobic ponds. Some designs rely on algae to provide oxygen for bacterial breakdown of organic matter. Sludge is not returned |
| Anaerobic digestion | Used for high strength organic effluents (e.g. pharmaceutical, food and drink industries). Waste water is stored in a sealed tank which excludes oxygen. Anaerobic bacteria breakdown organic matter into methane, carbon dioxide and organic acids. Final effluent still requires further treatment as has a high BOD. Also used for the stabilisation of sewage sludge at a concentration of 2–7% solids |

The volume of wastewater varies from country to country depending on its standard of living and the availability of water supplies (Table 1.9). Generally, the volume and strength of the sewage discharged in a particular country can be predicted fairly accurately. For example, the mean daily volume of wastewater, excluding industrial waste but including infiltration,

Table 1.9. Specific water consumption in Europe (IWSA 1995).

| | Household and small businesses | | Industry and others | | Total | |
|---|---|---|---|---|---|---|
| | 1980 | 1993 | 1980 | 1993 | 1980 | 1993 |
| Austria | 155 | 170 | 100 | 92 | 255 | 262 |
| Belgium | 104 | 120 | 59 | 37 | 163 | 157 |
| Denmark | 165 | 155 | 96 | 74 | 261 | 229 |
| France | 109 | 157 | 58 | 58 | 167 | 215 |
| Germany[1] | 137 | 136 | 74 | 41 | 211 | 177 |
| Hungary | 110 | 121 | 107 | 63 | 217 | 184 |
| Italy | 211 | 251 | 69 | 78 | 280 | 329 |
| Luxembourg | 183 | 178 | 76 | 83 | 259 | 261 |
| Netherlands | 142 | 171 | 37 | 32 | 179 | 203 |
| Norway | 154 | 180 | 247 | 340 | 401 | 520 |
| Spain | 157 | 210 | 58 | 90 | 215 | 300 |
| Sweden | 195 | 203 | 120 | 73 | 315 | 276 |
| Switzerland | 229 | 242 | 163 | 120 | 392 | 362 |
| United Kingdom | 154 | —[2] | 100 | —[2] | 254 | 331 |

[1] Includes former GDR.

[2] UK values not available in this format.

produced per capita in England is 180 l $d^{-1}$, compared 230 l $d^{-1}$ in Ireland and 250 l $d^{-1}$ in Scotland. The equivalent volume of sewage produced in the USA is on average 300 l per capita per day (100 US gallons $d^{-1}$). The amount of wastewater produced per capita can be estimated quite accurately from the specific water consumption.

The variation in volume depends on a number of variables including the amount of infiltration water entering the sewer. The higher volume of wastewater produced in Scotland is primarily due to the widescale use of a larger flushing cistern, 13.6 l compared with 9.0 l in England and Wales, although other factors also contribute to this variation. Guidelines from the Department of the Environment in England and Wales stipulate that all new cisterns manufactured after 1993 should have a maximum flushing volume of 7.5 l. However, the reliance of water closets which function on a siphon rather than a valve to release water restricts the minimum operational volume to between 4–5 l (Pearse 1987). The Building Research Establishment (1987) highlights the potential water saving from the adoption of new cistern designs and suggests the need for new British Standards.

Comparative studies were carried out using a 'standard turd', which is a 43 mm diameter ball of non-absorbent material with a relative density of 1.08, and with a cohesive shear strength, coefficient of friction, and adhesive properties very close to the real thing.

In rural areas, where water is drawn from boreholes or from small community water schemes, water may be at a premium, so the necessary conservation of supplies results in reduced volumes of stronger sewage. Occasionally, the water pressure from such rural supplies is too low to operate automatic washing machines or dishwashers and results in an overall reduction in water usage and subsequent wastewater discharge.

In the home, wastewater comes from three main sources. Approximately a third of the volume comes from the toilet, a third from personal washing via the wash basin, bath, and shower, and a third from other sources such as washing up, laundry, food and drink preparation (Tables 1.10 and 1.11). Outside the home, the strength and volume of wastewater produced per capita per day will fluctuate according to source, and this variation must be taken into account when designing a new treatment plant. For example, the flow per capita can vary from 50 l d$^{-1}$ at a camping site to 300 l d$^{-1}$ at a luxury hotel (Table 1.11). More detailed tables of the volume of wastewater produced from non-industrial sources, including the strength of such wastewater, are given by Hammer (1999) and also by Metcalf and Eddy (1991).

The diluted nature of wastewater has led to the development of the present system of treatment found in nearly all the technically-developed countries, which is based on treating large volumes of weak wastewater. In less developed communities, the high solids concentration of the waste

Table 1.10. Comparison of the percentage consumption of water for various purposes in a home with an office; indicating the source and make-up of wastewater from these types of premises (Mann 1979).

| Home (sources) | Total water consumed (%) | Office (sources) | Total water consumed (%) | |
|---|---|---|---|---|
| WC flushing | 35 | WC flushing | 43 | 63 |
| Washing/bathing | 25 | Urinal flushing | 20 | |
| Food preparation/drinking | 15 | Washing | 27 | |
| | | Canteen use | 9 | |
| Laundry | 10 | Cleaning | 1 | |
| Car washing/garden use | 5[a] | | | |

[a] May not be disposed to sewer.

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Table 1.11. Daily volume of wastewater produced per capita from various non-industrial sources (Mann 1979).

| Source category | Volume of sewage (litres/person/day) |
|---|---|
| Small domestic housing | 120 |
| Luxury domestic housing | 200 |
| Hotels with private baths | 150 |
| Restaurants (toilet and kitchen wastes per customer) | 30–40 |
| Camping site with limited sanitary facilities | 80–120 |
| Day schools with meals service | 50–60 |
| Boarding schools: term time | 150–200 |
| Offices: day work | 40–50 |
| Factories: per 8 hour shift | 40–80 |

Table 1.12. Comparison of the concentration of various compounds reported in urban runoff with precipitation, strictly surface runoff from roads and with combined sewer overflow (Pope 1980). All units are in mg $l^{-1}$ unless specified. Those marked with † are in mg $kg^{-1}$ and ‡ in kg curb $km^{-1}$.

| | Reported concentration range (mg $l^{-1}$) | | | |
|---|---|---|---|---|
| Parameter | Precipitation | Road/street runoff | Urban runoff | Combined sewer overflow |
| COD | 2.5–322 | 300 | 5–3100 | 93–2636 |
| BOD | 1.1 | 25–165 | 1–700 | 15–685 |
| Total solids | 18–24 | 474–1070 | 400–15322 | 150–2300 |
| Volatile total solids | — | 37–86 | 12–1600 | — |
| Suspended solids | 2–13 | 11–5500 | 2–11300 | 20–1700 |
| Volatile suspended solids | 6–16 | 100–1500 | 12–1268 | 113 |
| Settleable solids | — | — | 0.5–5400 | — |
| Total dissolved solids | — | 66–33050 | 9–574 | — |
| Volatile dissolved solids | — | 1630 | 160 | — |
| Conductance ($\mu$mho $cm^{-1}$) | 8–395 | 10000 | 5.5–20000 | — |
| Turbidity (JTU) | 4–7 | — | 3–70 | — |
| Colour (Pt-Co units) | 5–10 | — | 5–160 | — |
| Total organic carbon | 1–18 | 5.3–49 | 14–120 | — |
| Total inorganic carbon | 0–2.8 | — | 1.17 | — |
| Oils/hydrocarbons | — | 28–400 | 0–110 | — |
| Phenols | — | — | 0–10 | — |

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Table 1.12. (*Continued*)

| Parameter | Reported concentration range (mg $l^{-1}$) Precipitation | Road/street runoff | Urban runoff | Combined sewer overflow |
|---|---|---|---|---|
| Total nitrogen N | 0.5–9.9 | 0.18–4.0 | 1.1–6.2 | 4.0–63.3 |
| Organic N | 0.1–0.32 | 0.18–3.23 | 0.1–16 | 1.5–33.1 |
| Inorganic N | 0.69 | — | 1.0 | — |
| Ammonia N | 0.01–0.4 | 1–2 | 0.1–14.0 | 0.1–12.5 |
| Nitrate N | 0.02–5.0 | 0.31–2.62 | 0.1–2.5 | — |
| Nitrite N | 0–0.1 | — | 0-1.5 | — |
| Total phosphorus | 0.001–0.35 | 0.3–0.7 | 0.09–4.4 | 1.0–26.5 |
| Hydrolysable phosphorus | 0.8–0.24 | — | 0.1–10 | — |
| Aldrin | — | — | 'trace' | — |
| Dieldrin | 0.003† | $6.8 \times 10^{-6}$‡ | 'trace' | — |
| $p,p'$-DDD | — | $18.9 \times 10^{-6}$‡ | — | — |
| $p,p'$-DDD | — | $17.2 \times 10^{-6}$‡ | — | — |
| Heptachlor | 0.04† | — | 'trace' | — |
| Lindane | — | — | 'trace' | — |
| PCB | — | $311 \times 10^{-6}$‡ | — | — |
| Bromide | — | — | 5 | — |
| Chloride | 0.1–1.1 | 4–70000 | 2–25000 | — |
| Cadmium | 0.013–0.056 | 0.002–0.01 | 0.006–0.045 | — |
| Chromium | 0.023–0.08 | 0.018–1.0 | 0.01–27.0 | — |
| Copper | 0.06–0.48 | 0.007–2.55 | 0.041–0.45 | — |
| Iron | 0–3.05 | 5–440 | 0–5.3 | — |
| Lead | 0.024–10.4 | 1–113 | 0.01–14.5 | — |
| Mercury | — | 0.029 | — | — |
| Nickel | — | 0.02–1.5 | — | — |
| Zinc | 0.02–4.9 | 1–15 | 0.01–5.23 | — |
| Total coliform ($ml^{-1}$) | — | — | 240–99100 | — |
| Total coliform (organisms $km^{-1}$) | — | $15.9 \times 10^{10}$ | — | — |
| Faecal coliform ($ml^{-1}$) | — | — | 5500–11200 | — |
| Faecal coliform (organisms $km^{-1}$) | — | $0.9 \times 10^{10}$ | — | — |
| Faecal streptococcus ($ml^{-1}$) | — | — | 120–20000 | — |

makes it difficult to move to central collection and treatment sites, while the more diluted wastewater flows easily through pipes, and can be transported easily and efficiently via a network of sewers to a central treatment works. In isolated areas or underdeveloped countries, human waste is normally treated on-site, due to its smaller volume and less fluid properties (Feachem and Cairncross 1993; Mara 1996).

The collection and transport of sewage to the treatment plant is via a network of sewers. Two main types of sewerage systems are used, combined and separate. Combined sewerage systems are common in most towns in Britain. Surface drainage from roads, paved areas, and roofs are collected in the same sewer as the foul wastewater and piped to the treatment works. This leads to fluctuations in both the volume and the strength of sewage due to rainfall, and although the treatment works is designed to treat up to three times the dry weather flow of wastewater (DWF), problems arise if the rainfall is either heavy or continuous. During such periods, the wastewater becomes relatively diluted and the volume too great to be dealt with by the treatment works. Excess flow is, therefore, either directly discharged to a watercourse as storm water or stored at the treatment works in storm water tanks. The stored wastewater can be circulated back to the start of the treatment works once capacity is available. However, once the tanks become full, and then the settled wastewater passes into the river without further treatment where the watercourse, already swollen with rainwater, can easily assimilate the diluted wastewater because of the extra dilution now available.

A separate sewerage system overcomes the problem of fluctuations in sewage strength and volume due to rain, by collecting and transporting only the foul wastewater to the treatment works, and surface drainage is discharged to the nearest water course. Such systems are common in new towns in Britain and are mandatory in Canada and the USA. This type of sewerage system allows more efficient and economic treatment works to be designed as the variation in the volume and strength of the wastewater is much smaller and can be more accurately predicted. A major drawback with separate systems is that the surface drainage water often becomes polluted. All stormwater is contaminated to some degree because of contact during the drainage cycle: it passes over paved areas along roadside gullies to enter the sewer via a drain with a gully pot, which catches and removes solids that might otherwise cause a blockage in the sewer pipe (Bartlett 1981). The quality of urban runoff is extremely variable and biochemical oxygen demand (BOD) values have been recorded in excess of 7,500 mg $l^{-1}$ (Mason 1991; Lee and Bang 2000). It is the first flush of storm water that is particularly polluting as it displaces the anaerobic wastewater, rich in bacteria, that has been standing in the gully pots of the roadside drains since the last storm (Butler and Memon 1999). The runoff from roads is rich in grit, suspended solids, hydrocarbons including polycyclic aromatic hydrocarbons (Krein and Schorer 2000), heavy metals, pesticides such as the herbicide atrazine (Appel and Hudak 2001), and, during the winter, chloride from road-salting operations. Surprisingly, it also contains organic matter, not only in the form of plant debris such as leaves and twigs, but also dog faeces (Table 1.12). It has been estimated that up to 17 g $m^{-2}$ $y^{-1}$ of dog faeces are deposited onto urban paved areas and that the dog

Table 1.13. Chemical characteristics of treated effluents from three UK sewage treatment plants.

| | Source | | |
|---|---|---|---|
| Constituent[a] | Stevenage | Letchworth | Redbridge |
| Total solids | 728 | 640 | 931 |
| Suspended solids | 15 | | 51 |
| Permanganate value | 13 | 8.6 | 16 |
| BOD | 9 | 2 | 21 |
| COD | 63 | 31 | 78 |
| Organic carbon | 20 | 13 | |
| Surface-active matter | | | |
| Anionic (as Manoxol OT) | 2.5 | 0.75 | 1.4 |
| Non-ionic (as Lissapol NX) | | | 0.4 |
| Ammonia (as N) | 4.1 | 1.9 | 7.1 |
| Nitrate (as N) | 38 | 21 | 26 |
| Nitrite (as N) | 1.8 | 0.2 | 0.4 |
| Chloride | 69 | 69 | 98 |
| Sulphate | 85 | 61 | 212 |
| Total phosphate (as P) | 9.6 | 6.2 | 8.2 |
| Sodium | 144 | 124 | |
| Potassium | 26 | 21 | |
| Total hardness | 249 | 295 | 468 |
| pH value | 7.6 | 7.2 | 7.4 |
| Turbidity (ATU)[b] | | | 66 |
| Colour (Hazen units) | 50 | 43 | 36 |
| Coliform bacteria (no./ml) | 1300 | | 3500 |

[a] Results are given in mg $l^{-1}$, unless otherwise indicated.

[b] Absorptiometric turbidy units

population of a city the size of Manchester will produce an organic load equivalent to the human population of a small town of 25–30,000 people. In New York, the dog population deposits over 68,000 kg of faeces and 405,000 l of urine onto the streets each day, much of which is washed by storm water into local streams and rivers (Feldman 1974). The degree of contamination of urban runoff during a specific storm depends on: (i) the intensity and duration of the rainfall; (ii) the length of the preceding dry period, which controls the build up of pollutants on roads and in the quality of water stored in gully pots and gutters; (iii) seasonal variations that occur in the rainfall pattern and temperature which affects the degradation of organic matter; including leaf fall and the use of grit and salt during the winter, and (iv) the effectiveness of local authorities to clean roadside gullies and gully pots (Helliwell 1979). Unlike drainage from land, runoff from roads and paved areas is very rapid due to the short length of surface water sewers. The contaminated wastewater, therefore, reaches the receiving watercourse very quickly and before the dry weather flow has increased, so that any pollutants entering will receive minimum dilution. Where there is an accidental or deliberate spillage of chemicals or noxious wastes on roads, or in private yards, serious pollution of receiving waters is bound to occur. However, with combined sewerage systems such spillages can be confined at the treatment works and recovered or treated before reaching the watercourse (Sec. 2.1.1). During storm events, it is possible for combined sewers in particular to become overloaded, leading to the operation of sewer overflow systems. Combined sewer overflows (CSOs) discharge a mixture of wastewater and surface runoff that causes severe pollution in receiving waters (Balmforth 1990; Field *et al.* 1994). Extensive work has been undertaken to reduce the number of storm water overflows within sewer networks and to reduce the amount of storm water entering the sewer by using interception systems such as swales, percolation areas, porous roads and wetlands (Field *et al.* 1994; Debo and Reese 1995; Shutes *et al.* 1997; Sieker 1998; Adams and Papa 2000).

It is common, in both separate and combined sewers, for water not discharged as wastewater to enter the sewer via joints and cracks in the pipework. Infiltration water is normally from ground water sources and can be especially high during periods of rainfall. Few estimates of the extent of the problem are available, although some studies have found infiltration to be as high as 80% of the total volume in badly deteriorated sewers. In the USA, it is estimated that a mean value is 70 $m^3d^{-1}$ per km of sewer (30,000 US gallons per day per mile of sewer) (Clarke *et al.* 1971), although Grace (1979) recorded mean values some 50% less. As groundwater is generally very clean, infiltration has the effect of diluting the strength of wastewater and at the same time increasing the volume requiring treatment.

The flow rate of wastewater to treatment works is extremely variable, and although such flows follow a basic diurnal pattern, each treatment works tends to have a characteristic flow pattern. This pattern is controlled by such factors as: the time taken for sewage to travel from households to the treatment works, which is itself a function of sewer length; the degree of infiltration; the presence of stormwater and the variability in the water consumption practices of communities (Gower 1980). Industrial inputs obviously have a profound effect on flow rates, and industrial practices such as discharging wastes after 8-hour shifts can completely alter the expected normal flow pattern to a treatment plant. The basic flow pattern for a domestic wastewater treatment plant is shown in Fig. 1.3 with the minimum flow normally occurring in the early hours of the morning when water consumption is lowest and the flow consists largely of infiltration water. Flow rate rapidly increases during the morning when peak morning water consumption reaches the plant, followed by a second peak in the

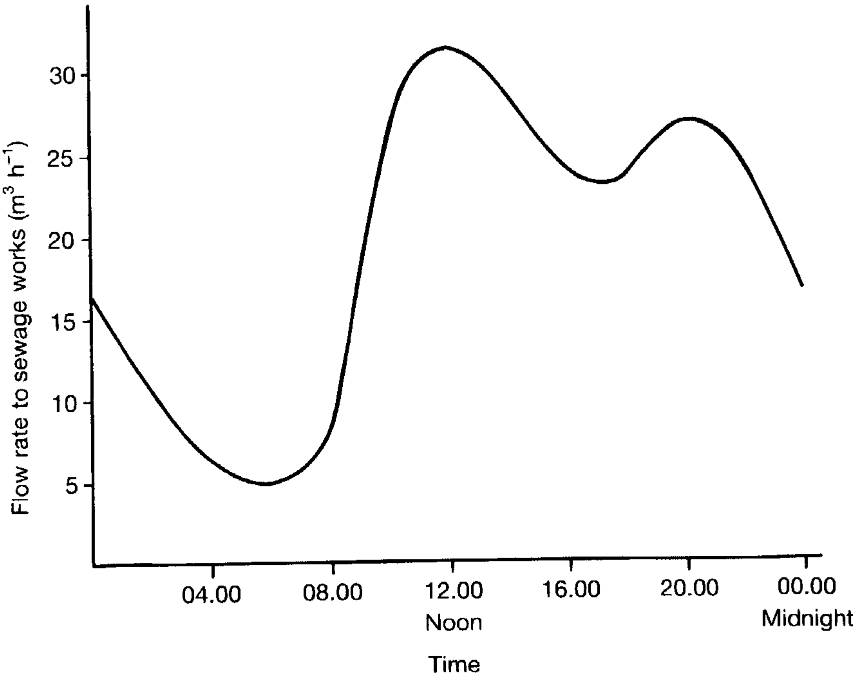

Fig. 1.3. Example of the hourly variation in flow to a sewage treatment plant.

early evening. When infiltration, storm water, and the water used for non-sewered purposes such as garden use, are removed from a basic model of consumption and discharge, then the water supplied is essentially equivalent to the wastewater discharged to the sewer (Lenz 1983a). Thus, the wastewater discharge curve, as measured at the sewage treatment works, will closely parallel the water supply curve, as measured at the waterworks, with a lag of several hours.

Infiltration and storm water tend to distort the basic shape of the hydrograph of diurnal flow. Infiltration, while increasing the total daily volume, does not alter its characteristic shape. Storm water, however, can alter the shape of the hydrograph by hiding peaks and troughs or adding new peaks as the rainfall causes rapid increases in the flow. Hourly fluctuations are less clear in large catchments due to the diversity of activities taking place during the 24-hour period and the presence of industry. The variable distance of households from the treatment works normally results in the hydrograph of the diurnal pattern becoming flattened and extended so that only one trough and one peak is seen daily (Clark *et al.* 1977; Escritt and Haworth 1984). Many problems at small to moderate sized treatment works are associated with the diurnal variation in flow, which is especially serious at the smallest works where often there is no flow at all during the night. Smaller variations of the average daily flow rate are recorded at treatment works serving large catchments (50–200%) compared with smaller communities (20–300%) (Painter 1958; Water Pollution Control Federation 1961). Many works overcome the problem of flow variation by using flow balancing, where the wastewater is stored at times of high flow and allowed to enter the works at a constant rate, or by recirculating treated final effluent during periods of low flow.

Variation between weekday flows is negligible, except in those areas where the household laundry is done on specific days. However, with the advent of automatic washing machines this practice has become largely extinct. With changing work patterns, many homes are now only occupied at night and on the weekends, leading to changes in diurnal and daily flow characteristics. Also, automatic washing of household laundry and dishes is increasingly done at night to take advantage of cheaper off-peak electricity tariffs. Although summer discharges normally exceed winter flows by 10–20%, up to 20–30% in the USA, seasonal variations in flow are due mainly to variation in population, as is the case at holiday resorts, schools, universities, and military camps. Other seasonal variations in flow are due to infiltration, which is linked to rainfall pattern and groundwater levels, and seasonal industrial activities such as food processing.

### 1.2.2. *Composition of sewage*

Wastewater is defined as domestic (sanitary) or industrial (trade). Domestic wastewater comes exclusively from residences, commercial buildings, and institutions such as schools and hospitals, while industrial wastewater comes from manufacturing plants. Inevitably, large towns and cities have a mixture of domestic and industrial wastewaters which is commonly referred to as municipal wastewater, and normally includes effluents from the service industries such as dairies, laundries, and bakeries, as well as a variety of small factories. It is unusual for modern municipal treatment plants to accept wastewater from major industrial complexes, such as chemical manufacturing, brewing, meat processing, metal processing, or paper mills, unless the treatment plant is specifically designed to do so. The practice in all European countries is now for water authorities to charge industry for the treatment and disposal of their wastewater. Thus, the current trend is for industry to treat its own waste in specifically designed treatment plants. In many cases, it is not cost-effective for an industry to provide and operate its own treatment plant, although most industries partially treat their waste to reduce the pollution load before discharge to the public sewer, in order to reduce excessive treatment charges.

It is of prime importance for the designer and operator of a treatment plant to have as much knowledge of the composition of the wastewater to be treated as possible. This is particularly important when new or additional wastes are discharged to existing plants. A full analysis of the wastewater will, for example:

(i) determine whether pretreatment is required;
(ii) determine whether an industrial waste should be treated alone or with sewage and, if so, in what proportions;
(iii) determine whether an industrial waste would attack the sewer;
(iv) permit a better selection of the most appropriate treatment process;
(v) allow an assessment of the toxicity or disease hazards;
(vi) provide indication of the resultant degree of eutrophication or organic enrichment in the form of sewage fungus in the receiving water (i.e. impact assessment); and
(vii) an assessment of the recoverable or reusable fractions of the wastewater.

Although there is considerable similarity in the basic content of sewage, the precise volume and characteristics will vary not only from country to country because of climatic conditions and social customs, but also within

Table 1.14. Volume and composition of human faeces and urine (Gloyna 1971).

| | Faeces | Urine |
|---|---|---|
| Moist weight per capita per day | 135–270 g | 1.0–1.3 kg |
| Dry weight per capital per day | 35–70 g | 50–70 g |
| Moisture content | 66–80% | 93–96% |
| Organic matter content (dry basis) | 88–97% | 65–85% |
| Nitrogen (dry basis) | 5.0–7.0% | 15–19% |
| Phosphorus (as $P_2O_5$) (dry basis) | 3.0–5.4% | 2.5–5.0% |
| Potassium (as $K_2O$) (dry basis) | 1.0–2.5% | 3.0–4.5% |
| Carbon (as dry basis) | 40–55% | 11–17% |
| Calcium (CaO) (dry basis) | 4–5% | 4.5–6.0% |

individual countries due to supply water characteristics, water availability, population size, and the presence of industrial wastes. Data on wastewaters is normally limited to $BOD_5$ (the five day biochemical oxygen demand test), COD (chemical oxygen demand), suspended solids, and ammonia, while a fuller characterisation of the wastewater being treated is rare (Tables 1.13 and 3.11). Analysis of wastewater composition can be done directly by laboratory examination of the sewage itself or indirectly by predicting the composition by examination of the gross components. Details of the composition of human faeces and urine are available (Table 1.14) although details of other household wastes which are more variable are less well-known, therefore, more direct methods of wastewater characterisation are preferred. Surprisingly, little is known of the composition of sewage and few specific studies have been carried out. Casanova *et al.* (2001) carried out a detailed study of the chemical and microbial characteristics of the wastewater generated by a single family home comprising two adults in Arizona in the USA. The wastewater is significantly weaker in terms of BOD and suspended solids compared with domestic wastewater treated at a central works. The wastewater contained high densities of total coliforms $8.03 \times 10^7$ CPU 100 $ml^{-1}$, faecal coliforms $5.63 \times 10^5$ CPU 100 $ml^{-1}$, faecal streptococci $2.38 \times 10^2$ CPU 100 $ml^{-1}$, and *Pseudomonas aeruginosa* $1.99 \times 10^4$ CPU 100 $ml^{-1}$. Legislation is setting even tighter controls on effluent quality, especially in terms of nutrients and listed substances (Table 1.4). More attention is now being paid to eliminating these at source rather than providing expensive and often energy intensive end of pipe solutions (Chap. 11). Some individual components of sewage which causes specific problems have been studied. For example, total phosphorus and nitrogen in eutrophication studies, detergents causing foaming, and indole in the control of odours.

Sewage is a complex mixture of natural inorganic and organic materials with a small proportion of man-made substances. The main source of pollution in sewage is human excreta with smaller contributions from food preparation, personal washing, laundry, and surface drainage. The chemical and physical nature of wastewaters can be further complicated by the inclusion of industrial wastes which are composed of strong spent liquors from main industrial processes and comparatively weak wastewaters from rinsing, washing and condensing.

The reason why sewage composition is normally measured in terms of $BOD_5$, COD, suspended solids, and ammonia content is because it is from these basic determinants that its polluting strength is assessed. Most charging systems are based on the Mogden formula, which uses these basic determinants. Other variables occasionally measured under specific circumstances, such as total phosphorus if the final effluent is discharged to inland lakes, or heavy metals if the sludge is to be subsequently used for agriculture. Charges are calculated from separate costs for reception, conveyance treatment and disposal actually given to the trade effluent. The basic formula is:

$$C = R + V + (Ot/Os)B + (St/Ss)S$$

where $C$ is the total charge in pence (sterling) per 1000 litres of trade effluent, $R$ is the reception and conveyance cost per 1000 litres, $V$ the volumetric and primary treatment costs per 1000 litres, $Ot$ the COD of trade effluent after one hour quiescent settlement (mg $l^{-1}$), $Os$ the COD of average strength settled sewage (mg $l^{-1}$), $B$ the cost of biological oxidation of settled sewage, $St$ the total suspended solids of the trade effluent (mg $l^{-1}$), $Ss$ the total suspended solids of average strength settled sewage (mg $l^{-1}$), and $S$ the treatment and disposal cost of primary sludge per 1000 litres of sewage.

Some companies have modified the basic Mogden formula to incorporate additional factors. These are $Bv$ or $Vb$, which is the additional volume charge if there is biological treatment (p $m^{-3}$), $M$ is cost for treatment and disposal where the effluent goes to a sea outfall (p $m^{-3}$), and $Md$ or $Vm$ a supplement to $M$ where the effluent goes to a designated long sea outfall (p $m^{-3}$). This gives a modified Mogden formula:

$$C = R + V([V + Bv] \text{ or } Vm \text{ or } M) + (Ot/Os)B + (St/Ss)S$$

Table 1.15. Trade charging formula employed in the UK during 1999/2000. Details of abbreviations are given in the text except: $Vb = V + Bv$; $Vm = M$; $Mo =$ monitoring costs; and $P =$ the cost per m$^3$ of the preliminary treatment required for foul sewage.

| Water service company | Charging |
|---|---|
| Anglian | $C = R + (V \text{ or } Vb \text{ or } Vm \text{ or } M) + (Ot/Os)B + (St/Ss)S$ |
| Dwr Cymru | $C = R + V \text{ or } Vb + (Ot/Os)B + (St/Ss)S$ |
| Northumbrian | $C = R + V + (Ot/Os)B + (St/Ss)S$ |
| North West | $C = R + V + M + B_1 + (Ot/Os)B_2 + (St/Ss)S$ |
| Severn Trent | $C = R + V + (Ot/Os)B + (St/Ss)S$ |
| Southern | $C = R + (V \text{ or } Vb \text{ or } Vm) + (Ot/Os)B + (St/Ss)S + M$ |
| South West | $C = R + V \text{ (or } Vm) + (Ot/Os)B + (St/Ss)S$ |
| Thames | $C = R + V + (Ot/Os)B + (St/Ss)S$ |
| Wessex | $C = R + V + (Ot/Os)B + (St/Ss)S$ |
| Yorkshire | $C = R + P(Ot/Os)B + (St/Ss)S$ |
| North of Scotland | $C = R + V + (Ot/Os)B + (St/Ss)S$ |
| East of Scotland | $C = R + V + (Ot/Os)B + (St/Ss)S + Mo$ |
| West of Scotland | $C = R + V + (Ot/Os)B + (St/Ss)S$ |
| Northern Ireland | $C = R + V + (Ot/Os)B + (St/Ss)S$ |

Specific company trade effluent charging formulae are given in Table 1.15, and charges in Table 1.16. Other variables are occasionally measured under specific circumstances, such as total phosphorus if the final effluent is disposed of to inland lakes or heavy metals if the sludge is to be subsequently used for agriculture.

The strength of sewage varies widely and depends on such factors as per capita water usage, infiltration, surface and storm water, and local habits. The water usage in the USA is at least three times greater than in Britain, which is why American sewage is usually weaker. Although the per capita production of organic matter is essentially the same in the USA and Britain, the difference in water consumption results in a raw sewage $BOD_5$ of between 100–700 mg $l^{-1}$ (with a mean $BOD_5$ of 320 mg $l^{-1}$) in Britain (Painter 1971). The use of garbage grinders or disposal units, so that household kitchen waste is disposed to the sewer rather than the refuse bin, results in a 30% increase in wastewater $BOD_5$ and 60% increase in the suspended solids. The concentration of nitrogen in domestic wastewater is directly related to the $BOD_5$, with about 40% of the total nitrogen in solution as ammonia. Proteins and urea undergo deamination releasing ammonia as the wastewater flows to the treatment plant. The longer the

Table 1.16. British trade effluent tariffs 2000–2001.

| | Minimum | | | | | | | Regional strengths | |
|---|---|---|---|---|---|---|---|---|---|
| Water & sewerage companies | charge £ | $R$ p/m$^3$ | $V$ p/m$^3$ | $Bv$ p/m$^3$ | $M$ p/m$^3$ | $B$ p/kg | $S$ p/kg | $Os$ mg/l | $Ss$ mg/l |
| Anglian — Green | 167.00 | 13.49 | 21.12 | 4.08 | 11.31 | 44.05 | 30.99 | 419 | 402 |
| — Orange | 167.00 | 12.58 | 19.69 | 3.80 | 10.55 | 41.08 | 28.92 | 419 | 402 |
| — Blue | 167.00 | 12.35 | 19.32 | 3.75 | 10.35 | 40.34 | 28.35 | 419 | 402 |
| — Industrial | 167.00 | 9.53 | 14.93 | 2.89 | 8.01 | 31.16 | 21.92 | 419 | 402 |
| Dŵr Cymru[1] | 112.50 | 17.29 | 19.68 | 8.18 | 11.77 | 25.55 | 26.41 | 500 | 350 |
| North West | 120.00 | 11.20 | 9.00 | 1.30 | 8.60 | 25.50 | 29.20 | 247 | 232 |
| Northumbrian | 230.00 | 18.66 | 9.15 | — | — | 33.20 | 37.72 | 386 | 187 |
| Severn Trent[2] | 86.60 | 12.53 | 11.81 | — | — | 20.11 | 15.35 | 351 | 343 |
| South West | 149.00 | 35.80 | 32.99 | — | 6.01 | 78.09 | 70.96 | 744 | 489 |
| Southern | 180.00 | 24.26 | 17.70 | 2.87 | 15.37 | 51.70 | 31.25 | 452 | 512 |
| Thames[3] | 71.00 | 6.71 | 8.25 | — | — | 24.95 | 35.87 | 445 | 336 |
| Wessex | 170.00 | 22.65 | 14.65 | — | 7.57 | 29.77 | 37.29 | 802 | 313 |
| Yorkshire[4] | 202.00 | 20.48 | 20.22 | — | 12.13 | 21.94 | 36.00 | 905 | 314 |

*Notes*:

[1] Dŵr Cymru has a reduced charge $R$ of 11.24p/m$^3$, where the volume discharged is $> 100,000$m$^3$ per annum. A fixed charge of £6,050 also applies to discharges $> 100,000$m$^3$.

[2] Severn Trent Water has a banded charge, $R \leq 49,999$m$^3$ charged at the standard rate of 12.53p/m$^3$, then $\geq 50,000$m$^3$ to $< 250,000$m$^3$ is charged at 11.49p/m$^3$ and $\geq 250,000$m$^3$ at 9.23p/m$^3$.

[3] Thames Water has a large user trade effluent tariff for customers with an annual bill > £58,000. This includes a fixed charge based on meter size, an annual charge of £10,000 and reception and treatment charges of R = 5.34p/m$^3$, $V$ = 6.56p/m$^3$, $B$ = 19.86p/m$^3$ and $S$ = 28.55p/m$^3$.

[4] Yorkshire Water has a banded charge, $R \leq 50,000$m$^3$ charged at the standard rate of 20.48p/m$^3$, then $> 50,000$m$^3$ to $\leq 250,000$m$^3$ is charged at 11.32p/m$^3$ and $> 250,000$m$^3$ at 7.72p/m$^3$. The $M$ charge is calculated as 60% of the $V$ charge.

Table 1.17. Comparison of typical chemical composition of raw wastewaters from the USA and UK.

| Parameter | USA (mg $l^{-1}$) | UK (mg $l^{-1}$) |
|---|---|---|
| pH | 7.0 | 7.2 |
| BOD | 250 | 326 |
| COD | 500 | 650 |
| TOC | 250 | 173 |
| Total solids | 700 | — |
| Suspended solids | 220 | 127 |
| Total nitrogen | 40 | 66 |
| Organic nitrogen | 25 | 19 |
| Ammonia nitrogen | 25 | 47 |
| Nitrite | 0 | 0 |
| Total phosphorus | 12 | 15 |
| Oraganic phosphorus | 2 | 3 |
| Inorganic phosphorus | 10 | 12 |

sewage is held in the sewer, the greater will be the release of ammonia. The per capita nitrogen production in the United Kingdom is 5.9 g N per day (Painter 1958) which is essentially the same as the American figure (Babbitt 1947). The per capita production of phosphorus is about a third of the weight of nitrogen produced, about 1.4 kg per capita per year. Of this, up to 70% comes from polyphosphate builders used in synthetic detergents (Table 1.17). Detergent polyphosphate builders are slowly being replaced by alternative phosphorus free builders such as zeolites.

The amount of organic matter produced per capita each day expressed in terms of $BOD_5$ is also known as the population or person equivalent. Population equivalent (PE), expressed in kg $BOD_5$ per capita per day, is determined as:

$$\mathrm{PE} = \frac{\text{mean flow (l)} \times \text{mean BOD}_5\ (\text{mg l}^{-1})}{10^6}$$

Population equivalent is often used in the design of treatment plants, and the volume and strength of industrial wastewaters are normally expressed in terms of equivalent population. In the UK, the PE of domestic sewage is equivalent to 0.055 kg $BOD_5$ per capita $d^{-1}$. This ranges from 0.045 kg for an entirely residential area to 0.077 kg for a large industrial city. American figures are similar for domestic sewage, being 0.052 kg for separate sewers and 0.063 kg for combined sewers. However, the recognised design figures in the USA and Canada are 0.077 kg $BOD_5$ and 0.10 kg suspended solids. Apart from $BOD_5$ and suspended solids, it is also common to quote total nitrogen or total phosphorus in terms of PE.

The PE of an industrial wastewater in Britain is calculated using the relationship:

$$\mathrm{PE} = \frac{\text{mean flow } (\mathrm{m^3 d^{-1}}) \times \text{mean } \mathrm{BOD_5} (\mathrm{mg\ l^{-1}})}{0.055 \times 10^3}$$

Similar to flow, the strength and composition of sewage changes on an hourly, daily, and seasonal basis. However, it is the diurnal variation that is usually the greatest. A similar diurnal pattern of sewage strength is formed, in terms of $BOD_5$ and suspended solids, as occurs with flow (Fig. 1.3). The peak in BOD occurs in the mid morning but, as with the flow, the actual time depends on the length of the sewers and the nature of the area served. The strength of sewage in large cities, with very long and complex sewerage systems, does not fluctuate as widely as it does in smaller catchments with maximum values occurring between 10 p.m. and 6 a.m. (Painter 1971). There is a wide diurnal variation in $BOD_5$ strength. This variation is also reflected by similar fluctuations in the concentrations of the various carbohydrates and fatty acids that largely make up the biodegradable carbon fraction (Painter 1958). Peaks also occur in the concentrations of ammonia and urea, occurring in the morning and late at night, reflecting the habits of the local population served. However, only the morning peak is generally discernible.

Strength varies from day to day due to the dilution effect of surface and storm water. The daily fluctuation in flow and strength is much less at treatment plants fed by a separate sewerage system. Wastewater volumes are greater on Mondays than any other day in the USA (Heukelekian and Balmat 1959), while the concentration of detergents in the UK has been reported as being higher on Mondays than the rest of the week (Eden and Truesdale 1961). However, this data was collected when automatic washing machines were not widely available and the household laundry for the week was done by necessity on a specific day, usually Monday, so it is unlikely that this trend is still discernible today. Seasonally, sewage strength does not vary significantly, although periods of drought and excessive rainfall can affect the dilution ratio in combined sewerage systems due to reduced infiltration and storm water. In the summer, bacterial concentrations in the wastewater reach a maximum, as do virus concentrations (Painter 1971).

Wastewater can only be treated biologically if sufficient nitrogen and phosphorus are present. Normally, there is a surplus of these nutrients in sewage for biological needs but it is necessary to assess the treatability of wastewaters by checking the ratio of carbon to nitrogen and phosphorus (C:N:P). The optimum C:N:P weight ratio for biological treatment is 100:5:1 (100 mg $l^{-1}$ $BOD_5$, 5 mg $l^{-1}$ N, 1 mg $l^{-1}$ P). The C:N:P ratio for raw sewage is approximately 100:17:5 and 100:19:6 for settled sewage, both containing abundant nutrients for microbial growth. The nitrogen requirement of the micro-organisms in the biological unit of a treatment plant is satisfied if the ratio of carbon, measured as $BOD_5$, to nitrogen equals or is less than 18:1. Even at C:N ratios $> 22:1$ removal still occurs, but much less efficiently.

Micro-organisms require much lower levels of phosphorus compared with nitrogen so the phosphorus requirement will be met if the C:P ratio is less than 90–150:1. Above 150:1, there is an increasing loss of efficiency (Porges 1960, Hattingh 1963, Komolrit, *et al.* 1967). The exact C:N:P ratio for optimum biological growth depends on the biological process and the form in which the nutrients are available in the wastewater. Where wastes fail to meet the C:N:P criteria, it becomes necessary to add nutrients in order to ensure that biological oxidation occurs. This is often carried out by mixing nutrient deficient wastes with sewage in the correct proportions (Sec. 3.3.2). Wastewaters from the brewing and canning industries are particularly deficient in nitrogen and phosphorus, therefore nutrient addition is required if optimum carbonaceous oxidation is to be achieved. An example of nutrient deficient wastewater is cited by Jackson and Lives (1972) who found that the BOD removal during the treatment of a cider effluent using low-rate percolating filtration was increased from 92 to 99% when the nitrogen and phosphorus balance was corrected by the addition of an inorganic supplement.

### *Physical properties*

Those who have not actually come into contact with raw sewage often harbour rather strange ideas as to what it looks and smells like. By the time it reaches the sewage treatment plant the vast majority of the large solids, such as faeces and paper, have broken up into very small particles. Thus, apart from a small quantity of floatable material, raw sewage is a rather turbid liquid with visible particles of organic material that readily settle out of suspension. The colour is normally grey to yellow-brown, according to the time of day. However, if all the oxygen has been used up during transit in the sewer then the wastewater becomes anaerobic or septic and takes on a much darker colour, and in extreme cases turns black. Municipal wastewaters receiving industrial wastes containing dyes will take on the colour of the dye present, and at treatment plants receiving effluents from the textile industry in particular, the raw wastewater undergoes spectacular and frequent colour changes. Under ultraviolet light, domestic sewage has a characteristic coloured fluorescence which is due to a variety of minor constituents present in household detergents.

Generally, domestic wastewater has a musty or earthy smell which is not at all offensive, although pungent odours can be produced if the wastewater becomes anaerobic. Certain industrial wastes and contaminants in surface runoff do have distinctive odours. Like a wine connoisseur, the ability of the operator to develop a discerning nose can be extremely helpful in identifying potential problems within the treatment plant and to identify changes in the composition of the sewage entering the plant. The odours produced are usually caused by gases produced by the decomposition of various fractions of the organic matter present in the wastewater. The commonest odour encountered is the smell of rotten eggs caused by hydrogen sulphide, which is produced by anaerobic bacteria reducing sulphate to sulphide. Table 1.18 lists the major categories of odours encountered at sewage treatment plants, although the quantification of such odours for use as a control variable has proved extremely difficult (American Public

Table 1.18. Some characteristic odours produced by compounds present in wastewaters. These degradation products can be categorised into two main groups, either degradation products of nitrogenous or sulphurous compounds. There are other odourous compounds such as those associated with chlorine and phenolic wastes.

| Compounds | General formulae | Odour produced |
|---|---|---|
| *Nitrogenous* | | |
| Amines | $CH_3NH_2$, $(CH_3)_3N$ | Fishy |
| Ammonia | $NH_3$ | Ammoniacal, pungent |
| Diamines | $NH_2(CH_2)_4NH_2$, $NH_2(CH_2)_5NH_2$ | Rotten flesh |
| Skatole | $C_8H_5NHCH_3$ | Faecal, repulsive |
| *Sulphurous* | | |
| Hydrogen sulphide | $H_2S$ | Rotten eggs |
| Mercaptans | $CH_3SH$, $CH_3(CH_2)_3SH$ | Strong decayed cabbage |
| Organic sulphides | $(CH_3)_2S$, $CH_3SSCH_3$ | Rotten cabbage |
| Sulphur dioxide | $SO_2$ | Pungent, acidic |
| *Other* | | |
| Chlorine | $Cl_2$ | Chlorine |
| Chlorophenol | $Cl.C_6H_4OH$ | Medicinal, phenolic |

Health Association *et al.* 1983; Metcalf and Eddy Inc. 1991). Some food processing wastewaters produce extremely strong odours, especially during treatment and storage (Gerick 1984; Gray 1988). For example, sugar beet wastewater undergoes partial anaerobic breakdown within the process and subsequently on storage in lagoons, with the production of a variety of odours. The major odours come from the volatile fatty acids that comprise most of the organic fraction of the effluent. The odour threshold concentrations for the volatile acids produced during treatment of sugar beet are 24.3 ppm for acetic acid, 20.0 ppm propionic acid, 0.05 ppm iso-butyric acid, 0.24 ppm butyric acid, 0.7 ppm iso-valeric acid, and 3.0 ppm for valeric acid. Therefore, by measuring the volatile acid concentration and dividing it by the appropriate odour threshold concentration, a measure of the odour production or concentration known as the odour number can be calculated (Gray 1988). Other odours associated with sugar beet processing include trimethylamine which has a fishy odour and organic sulphides which produce a strong odour of rotting cabbage, as do the thiol compounds methyl mercaptan ($CH_3SH$) and ethyl mercaptan ($C_2H_5SH$), both of which have very low odour thresholds of 0.0011 ppm and 0.00019 ppm respectively (Shore *et al.* 1979). A detailed list of odour threshold values has been compiled by Fazzalari (1978).

The temperature of sewage is normally several degrees warmer than the air temperature, except during the warmest months, due to the specific heat of water being much greater than that of air. Sewage temperatures are normally several degrees warmer than the water supply and because of its high conductivity rarely freezes in temperate climates. In the UK, the temperature of raw sewage ranges from 8–12°C in winter to 17–20°C in the summer (Painter 1971). Comparative studies have shown that the variability in the temperature of settled sewage, as it enters the percolating filtration stage at a treatment works in South Yorkshire, is 30–50% less than that recorded for the air temperature, a total range of 11.5°C and 30.4°C respectively. Both the sewage and air temperatures followed similar seasonal patterns, reaching maximum and minimum temperatures during the same periods. However, while the mean daily air temperature varied annually, the mean daily temperature of the sewage remained constant at 12.4°C (Gray 1980). In hotter climates domestic sewage can be much warmer, and in India sewage temperatures of 28–30°C are not unusual (Kothandaraman *et al.* 1963).

The pH of sewage is usually above 7 with the actual value depending largely on the hardness of the supply water. Although extreme values are occasionally encountered, soft water catchments generally have a pH range of 6.7–7.5 (modal pH = 7.2) and hard water catchments a range of 7.6–8.2 (modal pH = 7.8) (Painter 1971).

Total solids (i.e. the weight of matter remaining after a known volume of wastewater has been evaporated at 105°C) is a commonly used wastewater variable in the USA. It provides a simple characterisation of the wastewater with which the theoretical performance of unit treatment processes can be predicted. Total solids can be classified as either suspended or filterable depending on whether solids will pass through a standard filter (Department of the Environment 1972).

The standard filter used in Britain is a Whatman GF/C filter paper, the GF referring to its glass fibre structure, which has a pore size of 1.0 $\mu$m. Therefore, all the filterable solids have a particle size of $< 1.0$ $\mu$m. The suspended solids fraction ranges from colloidal particles $< 1.0$ $\mu$m up to recognisable gross matter. A portion of the suspended solids fraction is settleable, which is measured by measuring the volume of solids that settle out of suspension over a 60 minute period under quiescent conditions. An Imhoff cone, which is an inverted 1 litre conical flask, is used for this purpose and provides a useful estimate of the solids removal and sludge production during primary sedimentation. The filterable fraction contains colloidal and dissolved material. The colloidal solids are particulate ranging from 1 nm–1 $\mu$m in size, which are too small to be removed by gravity settlement. An assessment of colloidal solids can be made by measuring the light-transmitting properties of the wastewater, the turbidity, as colloidal matter scatters and absorbs light. The dissolved fraction is made up of both organic and inorganic molecules that are in solution.

Each of these major categories of solids is comprised of both organic and inorganic material, and the ratio of each can be measured by burning off the organic fraction in a muffle furnace at 600°C (Allen 1974). Certain salts are also destroyed by heating, although only magnesium carbonate is decomposed at this temperature being transformed to magnesium oxide and carbon dioxide at 350°C. The major inorganic salt in domestic sewage is calcium carbonate, but this remains stable up to 825°C. The percentage of each solids fraction in a wastewater depends on the chemical composition of the sewage. However, Painter (1971) separated the particulate solids in domestic sewage as approximately 50% settleable ($> 100$ $\mu$m diameter), 30–70% supra-colloidal (1–100 $\mu$m) and the remaining 17–20% as colloidal (1 nm–1 $\mu$m). However, a more detailed breakdown of the proportion of particular solids fractions in domestic wastewater and the organic strength of each fraction is shown in Table 1.19.

Table 1.19. The organic strength in terms of chemical oxygen demand (COD) of the various solids fractions of sewage (adapted from Rickert and Hunter 1971).

| | Total solids | | Organic content | | COD | |
|---|---|---|---|---|---|---|
| Solids fraction | mg $l^{-1}$ | % | mg $l^{-1}$ | % | mg $l^{-1}$ | % |
| Settleable | 74 | 15 | 59 | 25 | 120 | 29 |
| Supra-colloidal | 57 | 11 | 43 | 18 | 87 | 21 |
| Colloidal | 31 | 6 | 23 | 9 | 43 | 10 |
| Soluble | 351 | 68 | 116 | 48 | 168 | 40 |
| Total | 513 | — | 241 | — | 418 | — |

### *Organic properties*

Organic matter comprises of carbon, hydrogen and oxygen with nitrogen frequently present. Sulphur, phosphorus, and iron are only occasionally present. In medium strength sewage, 75% of the suspended solids and 40% of the filterable solids fractions are organic. In settled sewage, Painter (1983) estimates that 50% of the organic carbon and between 35–50% of the organic nitrogen is in solution. While three-quarters of the organic carbon can be attributed to the major organic groups carbohydrates, fats, proteins, amino acids and volatile acids, the remainder is made up of other organic molecules such as hormones, vitamins, surfactants, antibiotics, hormonal contraceptives, purines, pesticides, hydrocarbons and pigments. Many of the synthetic organic molecules are non-biodegradable while others are only decomposed biologically at very slow rates. The organic constituents of suspended solids and the filterable fraction of sewage are very different (Table 1.20). Carbohydrates are the largest group in solution in British sewage, with non-volatile and volatile acids, free and bound amino acids and anionic detergents all major constituents. Urea is a major component of urine but is hydrolysed so rapidly to ammonia that it is only found in very fresh sewage. The composition of sewage changes rapidly on storage due to bacterial action with the sugars in particular quickly converted to organic acids. Fats are the major organic constituents in the suspended solids fractions, and together with carbohydrates and proteins account for 60–80% of the organic carbon present.

Most of the naturally occurring amino acids, carbohydrates and organic acids are found in sewage. Of the carbohydrates, glucose, sucrose and lactose are the major ones with smaller proportions of galactose, fructose, xylose and arabinose. Together they account for 90–95% of all the carbohydrate

Table 1.20. Organic constituents of domestic sewage (Painter 1983).

| | In solution | | In suspension | |
|---|---|---|---|---|
| Constitutent | Concentration (mg $l^{-1}$) | Proportion as C of total C in solution (%) | Concentration (mg $l^{-1}$) | Proportion as C of total C in suspension (%) |
| Fats | — | — | 140 | 50 |
| Carbohydrates | 70 | 31.3 | 34 | 6.4 |
| Free and bound amino acids | 18 | 10.7 | 42 | 10 |
| Volatile acids | 25 | 11.3 | 12.5 | 2.3 |
| Non-volatile acids | 34 | 15.2 | | |
| Detergents (ABS) | 17 | 11.2 | 5.9 | 1.8 |
| Uric acid | 1 | 0.5 | — | — |
| Creatine | 6 | 3.9 | — | — |
| Amino sugars | — | — | 1.7 | 0.3 |
| Amides | — | — | 2.7 | 0.6 |
| Organic carbon | | | | |
| by direct analysis | 90 | 100 | 211 | 100 |
| by addition of above constituents | 75.6 | 84.1 | 151 | 71.4 |

present which is equivalent to 50–120 mg $l^{-1}$. A diurnal variation in carbohydrate concentration and composition is evident, and although glucose accounts for over 50% of the total carbohydrate content in composite samples, sucrose concentration is greater than glucose in the afternoon. The ratio of hexose to pentose is between 10 and 12. The non-soluble high molecular weight carbohydrates such as starches, cellulose and wood fibre are restricted to the suspended solids fraction resulting in a low hexose to pentose ratio (2.0–2.6) and a concentration of 30–38 mg $l^{-1}$.

In wastewater terminology, fats is a general term as is lipids or grease, to describe the whole range of fats, oils, and waxes discharged to the sewer. They are among the more stable organic compounds and are not easily degraded biologically. The major source of fats is from food preparation and to a lesser extent excreta, the major sources being butter, lard, margarine, vegetable fats and oil, meat, cereals, nuts and certain fruit. Fats are only sparingly soluble in water and so are only an important component of the suspended fraction of the wastewater, contributing up to 50% of the total carbon present. Normal concentration ranges for fats in domestic wastewater are between 40–100 mg $l^{-1}$, although this figure is normally higher than that recorded for American sewages. Fats are broken down by hydrolytic action to yield fatty acids and a wide variety of free fatty acids have been reported from sewage, including all the saturated ones from $C_8$ to $C_{14}$ as well as $C_{16}$, $C_{18}$ and $C_{20}$. The major acids include palmitic, stearic and oleic which form between 75–90% of those present. Full details of the fat content of domestic sewage are given by Painter (1971).

Acetic acid is the major volatile acid found in sewage, being recorded at concentrations between 6–37 mg $l^{-1}$, and together with propionic, butyric and valeric acids make up 90% of the total volatile acidity in wastewater. The acidity of sewage rapidly increases on storage at the expense of sugars and if high concentrations are recorded in fresh sewage, anaerobosis should be suspected. Non-volatile soluble acids are present at concentrations between 0.1–1.0 mg $l^{-1}$, the commonest being glutaric, glycolic, lactic, citric benzoic and phenyllactic acids.

Proteins are a comparatively important source of carbon in wastewater although they are less important than soluble carbohydrate or fats in suspension. Protein is the principal constituent of all animal and to a lesser extent plant tissue, so waste from food preparation and excreta are both rich in protein such as casein from milk or albumen and gelatine from animal tissue and bone. Apart from containing carbon, hydrogen and oxygen, proteins also contain a fairly high proportion of nitrogen, which is consistent at about 16%. Proteins, apart from urea, are the chief source of nitrogen in wastewater and supply up to 80% of the total organic nitrogen present. Proteins are made up of long chains of amino acids connected by peptide bonds, and are readily broken down by bacterial action to form free amino acids, fatty acids, nitrogenous compounds, phosphates and sulphides. In wastewater, the free amino acids generally account for $< 5$ mg N $l^{-1}$, although this can occasionally be higher, while bonded amino acids in the form of peptides or protein account for between 4–15 mg N $l^{-1}$.

Apart from amino acids, urea is also a major source of nitrogen in sewage, providing between 2–16 mg N $l^{-1}$. Urea is most abundant in fresh sewage as it is rapidly converted to ammonia under both aerobic and anaerobic conditions. The rate of conversion to ammonia has been estimated at 3 mg N $l^{-1}$ per hour at 12°C in stored samples (Painter 1958).

Not all biodegradable organic matter found in sewage can be classified into one of the major categories, and some natural compounds are in fact combinations of carbohydrates, proteins and fats such as lipoproteins and nucleoproteins. Of the organic matter in wastewater, 20–40% is essentially non-biodegradable within the treatment plant. However, the actual proportion of non-biodegradable material is normally very small. Fractions such as lignins and cellulose are only slightly degraded due to the limited time for decomposition within the treatment plant, and the absence of many of the specific micro-organisms required for decomposition in normal sewage.

Wastewater can contain an enormous range of specific compounds. It is estimated that some 50,000 dangerous chemicals are currently used within the EU, of which some 4,500 are potential List I substances (76/464/EEC). Therefore, where industrial wastewaters are discharged to sewer, they require careful characterisation and pretreatment as necessary (Howard 1990, 1991; EPA 1998). Although chemicals with a low vapour pressure, high adsorptivity onto solids, or a high solubility in water, are unlikely to vaporise and so become concentrated in the air space within the sewer, those chemicals showing the opposite characteristics are likely to vaporise and possibly cause explosions. Oestrogen mimicking compounds are currently causing much concern, as they have been shown to disrupt the endocrine systems of humans and wildlife (Dempsey 1998; Fawell and Chipman 2001). These compounds exhibit a wide chemical diversity and are defined more by their biological function than their composition. Natural animal oestrogen hormones (e.g. oestrone, 17ß - oestradiol) and synthetic oestrogen (ethinyl oestradiol) are the most oestrogenically active chemicals found in wastewater. Other Oestrogen mimicking compounds include: organochlorine pesticides (e.g. PCBs), organotins and dioxins. Alkylphenol ethoxylates (APEs) which are widely used in industrial detergents, paint formulations and metal finishing are also common. Some 18,000 tonnes of APEs are used annually in the UK; phytoestrogens and phytosterols which are naturally found in certain trees and plants and are strongly associated with wood pulp effluents; and finally, some phthalates which are ubiquitous in the environment. They are used as plasticers in adhesives, food wrapping and packaging. Bisphenol-A is a plastic manomer used in food and drink packaging, including the protective coating in food cans.

Sewage contains a diverse range of organisms which originate not only from faeces, but also from soil and water. They include viruses, bacteria, fungi, protozoans and a variety of other groups of organisms. Many of these organisms are pathogenic to man and for this reason are discussed fully in Chap. 9.

### *Inorganic properties*

There is a substantial inorganic component in sewage, especially compounds continuing sodium, calcium, potassium, magnesium, chlorine, sulphur (as sulphates and other forms), phosphates, bicarbonates, and ammonia. Traces of heavy metals are also found. The inorganic content of domestic wastewater depends on the geology of the catchment from which the water supply originated, (natural water dissolves minerals from the surrounding rocks and soil of the area), and on the nature of the polluting material itself. This is vividly illustrated by comparison of sewages from a soft and hard water area (Table 1.21). In hardwater catchments, the calcium, sodium, and chloride, ions are significantly more concentrated in the supply water, which is reflected in the resulting wastewater. Domestic wastewater contains a very wide range of inorganic salts and trace elements, including all those necessary for biological growth and activity. When trace elements are limited (e.g. Ca, Mg, Fe), then biological treatment efficiency will be reduced (Sec. 3.3.2). Among the major ions in wastewater which are worthy of further discussion are chloride, nitrogen, and phosphorus.

Chloride is found naturally in water due to leaching, but it also originates from a wide variety of agricultural, industrial and domestic sources. While infiltration by groundwater contaminated with salt-water into the sewer is a major source of chloride and sulphate in some areas, a major seasonal source at treatment plants served by combined sewerage systems is from road runoff during salting operations in icy weather. In hardwater areas, the widespread use of water softeners can result in significant increases in the wastewater chloride concentrations. Without these additional sources, sewage normally contains between 30–100 mg $l^{-1}$ of chloride:

Table 1.21. Concentration of major inorganic constituents of domestic sewage (Painter 1971).

| Constituent | Whole sewage USA Soft water area (mg $l^{-1}$) | Settled sewage UK Hard water area (mg $l^{-1}$) |
|---|---|---|
| Cl | 20.10 | 68.00 |
| Si | 3.90 | — |
| Fe | 0.80 | 0.80 |
| Al | 0.13 | — |
| Ca | 9.80 | 109.00 |
| Mg | 10.30 | 6.5 |
| K | 5.90 | 20.00 |
| Na | 23.00 | 100.00 |
| Mn | 0.47 | 0.05 |
| Cu | 1.56 | 0.2 |
| Zn | 0.36 | 0.65 |
| Pb | 0.48 | 0.08 |
| S | 10.30 | 22.00 |
| P | 6.60 | 22.00 |

human excreta contains 6 g Cl per capita per day and urine contains 1% chloride. As chloride is not removed to any great extent by conventional treatment, the detection of higher concentrations in surface waters may indicate that they are being used for wastewater disposal.

Nitrogen and phosphorus are both essential nutrients for plant growth. Nitrogen is also essential for the synthesis of protein and so biological growth generally. In fresh wastewater, nitrogen is primarily present as proteinaceous matter and urea. This organic nitrogen is rapidly decomposed by bacterial action in the case of proteins or by hydrolysis in the case of urea to ammonia, the concentration of which in wastewater is indicative to some extent of its age. Ammonia N exists in aqueous solution as either the ammonium ion ($NH_4^+$) or as ammonia ($NH_3$), depending on the pH of the wastewater. At pH values of $> 7$, the equilibrium of the reaction:

$$NH_3 + H_2O \longleftrightarrow NH_4^+ + OH^-$$

is displaced to the left so that ammonia predominates and at pH values $< 7$, equilibrium moves to the right and ammonium predominates (Fig. 1.4). Organic nitrogen is normally measured separately from ammonia, although occasionally they are expressed together as the kjeldhal nitrogen. The normal concentration range of nitrogen in settled sewage in Britain is 41–53 mg

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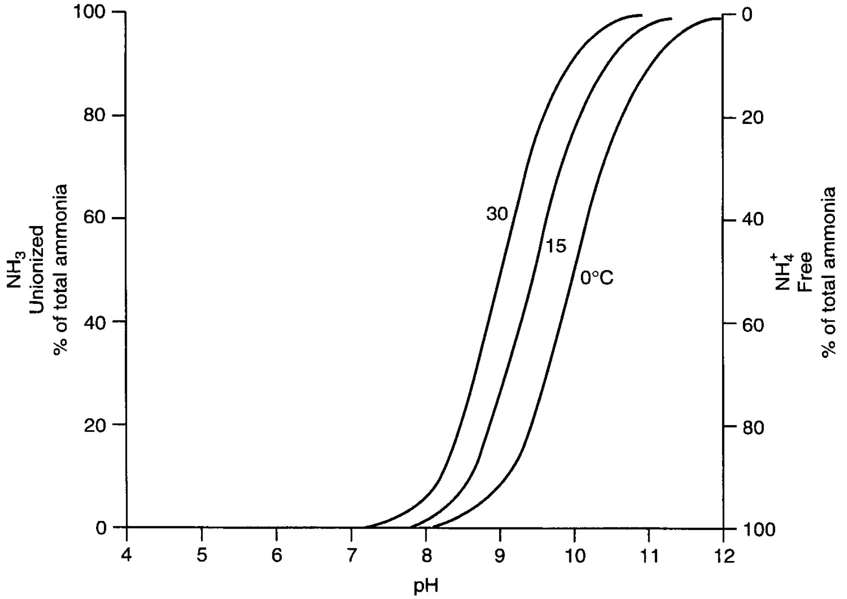


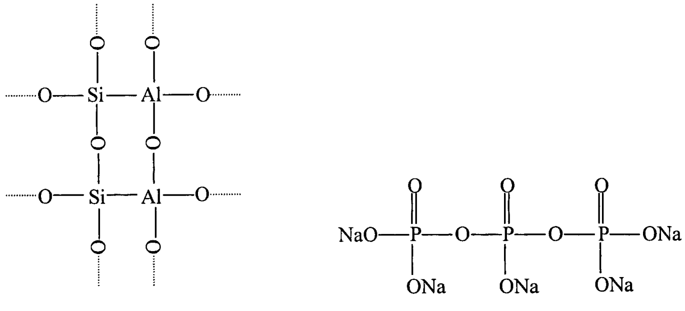
Fig. 1.4. The general variation between the proportion of unionised to free ammonia at varying pH and temperatures (Chapman 1996).

N $l^{-1}$ as ammonia, 16–23 mg N $l^{-1}$ as organic nitrogen and 57–76 mg N $l^{-1}$ as kjeldhal nitrogen (Painter 1971). The oxidised forms of ammonia, nitrite and nitrate, are normally absent from fresh sewage being products of the biological oxidation processes within the treatment plant. Therefore, as the total nitrogen includes all chemical forms of nitrogen, the kjeldhal nitrogen can be assumed to be equivalent to the total nitrogen in raw and settled sewage.

Phosphorus is present in sewage in three distinct forms, as orthophosphate, polyphosphate, and organic phosphate. Organic phosphate is a minor constituent of sewage and like the polyphosphates requires further decomposition to the more assimilable orthophosphate form, which is normally fairly slow. About 25% of the total phosphorus in settled sewage is present as orthophosphates, such as $PO_4^{3+}$, $HPO_4^{2-}$, $H_2PO_4^-$, $H_3PO_4$, which are available for immediate biological metabolism. Therefore, in terms of utilisation both in the treatment plant and subsequently in receiving waters, it is the inorganic phosphate concentration that is important rather than the total phosphorus concentration. After secondary treatment, about 80% of the total phosphorus in a final effluent is in the orthophosphate form. Average phosphorus concentrations in sewage range from 5–20 mg P $l^{-1}$

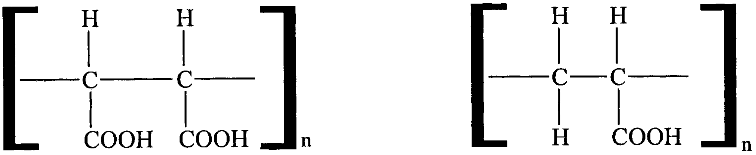
as total phosphorus, of which 1–5 mg P $l^{-1}$ is the organic fraction and the rest inorganic.

Since 1965, legislation in the USA and a voluntary ban in Britain has seen a steady reduction in the use of 'hard' or non-biodegradable alkyl-benzene-sulphonate (ABS) detergents in favour of 'soft' biodegradable linear-alkyl-sulphonate (LAS) detergents. The ABS detergents were responsible for persistent foaming problems at both sewage treatment plants and in receiving waters (Klein 1972b). Detergents are made up of a number of compounds, each with a specific function during the washing process (Broze 1999). All detergents vary in their specific formulation, although all generally contain the basic functional groups of compounds: surfactant (e.g. linear alkyl benzene sulphonate) 3–15%; builder (e.g. sodium tripolyphosphate) 0–30%; ion-exchanger (e.g. zeolite A) 0–25%; antiredeposition agent (e.g. polycarboxylic acids) 0–4%; bleaching agent (e.g. sodium perborate) 15–35%; bleach stabilizer (e.g. phosphonate) 0.2–1.0%; foam booster (e.g. ethanolamide) 1–5%; enzyme (e.g. protease) 0.3–1.0%; optical brightener (e.g. pyrazolan derivatives) 0.1–1.0%; corrosion inhibitor (e.g. sodium silicate) 2–7%; and a fragrance 0.05–0.3%. However, to increase the washing ability of LAS detergents, 'builders' in the form of sodium tripolyphosphate (STPP), are included. Their function is to remove hardness, which interferes



[Zeolite]

[Sodium tripolyphosphate]



[Polymaleic]

[Polyacrylic acid]

Fig. 1.5. Chemical structures of common detergent builders.

with the action of the surfactant, buffer the water to ensure optimum alkaline washing conditions, and to emulsify soils and prevent their redeposition. Sodium tripolyphosphate is made from phosphoric acid, neutralised with caustic soda or soda ash to make either monosodium orthophosphate ($NaH_2PO_4$) or disodium orthophosphate ($Na_2HPO_4$). The molecular water is then removed in a kiln to yield STPP ($Na_5P_5O_{10}$) (Fig. 1.5) (Morse *et al.* 1994). These particular phosphates are highly unstable and readily break down to super-phosphates within the treatment plant, which has resulted in growing problems of high phosphate concentrations in the final effluents being discharged to surface waters. In order to reduce the level of phosphates in the environment, certain countries have introduced legislation to control the use of STPP in detergent formulations. For example, the Irish Government have implemented a four year plan to phase out the use of phosphate-based detergents by the end of 2002 (DOELG 1999). However, phosphate comes from a variety of sources (Table 1.22) and it is unlikely that replacing detergent phosphates will solve eutrophication on its own (Morse *et al.* 1995). In Ireland, almost 75% of phosphorus inputs to lakes and rivers come from agriculture (EPA 2000). Initially, organic builders were used to replace STPP, the most efficient being nitrilotriacetic acid (NTA), polyacrylic acid (PAA), polymaleicacid (PMA) and polycarboxylic

Table 1.22. Phosphate sources in Europe in 1992 as the percentage from each source and total in tonnes per annum (Morse *et al.* 1993).

| Source (%) | Human | Detergents | Livestock | Fertilisers | Industry | Background | Total |
|---|---|---|---|---|---|---|---|
| **Member State** | | | | | | | |
| Austria | 20 | 10 | 36 | 16 | 6 | 12 | 13 |
| Belgium | 26 | 11 | 43 | 7 | 8 | 5 | 13 |
| Denmark | 12 | 11 | 55 | 11 | 5 | 6 | 15 |
| Finland | 18 | 9 | 17 | 15 | 3 | 38 | 9 |
| France | 18 | 15 | 31 | 19 | 6 | 11 | 106 |
| Germany | 28 | 3 | 44 | 12 | 6 | 7 | 97 |
| Greece | 21 | 7 | 18 | 34 | 5 | 15 | 17 |
| Ireland | 9 | 7 | 49 | 24 | 2 | 9 | 15 |
| Italy | 35 | 2 | 26 | 18 | 8 | 11 | 56 |
| Netherlands | 23 | 3 | 57 | 9 | 5 | 3 | 24 |
| Portugal | 24 | 14 | 27 | 16 | 7 | 12 | 15 |
| Spain | 19 | 16 | 18 | 26 | 7 | 14 | 72 |
| Sweden | 21 | 10 | 15 | 14 | 7 | 33 | 14 |
| UK | 24 | 19 | 29 | 14 | 8 | 6 | 82 |
| **EU Total** | 23 | 11 | 32 | 17 | 7 | 10 | 548 |

acids (PCAs) (Hunter *et al.* 1988) (Fig. 1.5). Although initially thought to be a benign alternative to STPP, problems about the toxicity of these organic builders were quickly identified (Anon 1994; Rand 1995). For that reason, zeolites have been widely adopted as an alternative builder to STPP, and are used in conjunction with polycarboxylates and sodium carbonate. Zeolite A ($Na{-}2OAl_2O_3 \cdot 2SiO_2 \cdot 4.5H_2O$) has a three dimensional framework of $AlO_4$ and $SiO_4$ tetrahedra linked by the oxygen atoms they share (Fig. 1.5). Its internal surface area is many times larger than the external surface area, allowing for greater sequestration of hardness ions than initially appears possible. However, it appears that zeolite based detergents are less efficient than STPP based detergents, with more of the former needed each wash to obtain the same performance as the latter detergent formulation (Wilson and Jones 1995). Zeolites increase the solids concentration in sewage sludge by up to 30% and improve activated sludge settling qualities; although, filterability deteriorates and the aluminium concentration increases. Overall zeolites do not appear to affect treatment efficiency or encourage filamentous bacterial development (Piirtola *et al.* 1998), although some environmental concerns have been raised (Wilson and Jones 1995).

Sulphur is another essential element in the metabolism of all organisms. However, most micro-organisms only require small amounts of the element which is used in the synthesis of the amino acids cysteine and methionine, which are found in protein. Nearly all the requirement for sulphur comes from sulphate. Trace quantities of all the metals necessary for biological growth are present in sewage. Those metals which are particularly toxic in excessive concentrations and which are common in sewage are nickel, lead, chromium, cadmium, zinc, and copper. Vacker *et al.* (1967), Heukelekian and Balmat (1959), and Painter (1958) have provided mean values of the most important metals in sewage and mean values using this data, which were collected from very different catchments, have been given by Gould (1976). This data, and metal concentrations collected from a domestic sewage treatment works in South Yorkshire, are compared in Table 1.23. The data from South Yorkshire has much lower concentrations of chromium and copper as it is a purely domestic sewage, however the high iron and lead concentrations are due to mine water infiltration and runoff from the nearby M1 motorway respectively. There has been a recent increase in the use of antiseptic creams and hair shampoos which are rich in zinc and this has led to a significant increase in the zinc concentration of sewage.

Sewage remains aerobic as long as it is not permitted to stand. Normal dissolved oxygen concentrations in flowing sewage are usually in the order of

Table 1.23. Variation of metals found in domestic sewage.

| Metal | Gould (1976) (mg $l^{-1}$) (mean) | Gray (1980) (mg $l^{-1}$) (range) |
|---|---|---|
| Cd | 0.02 | 0.00–0.05 |
| Co | $< 0.02$ | 0.00–0.01 |
| Cr | 0.4 | 0.00–0.1 |
| Cu | 0.88 | 0.00–0.16 |
| Fe | 0.8 | 0.15–1.30 |
| Mn | 0.2 | 0.01–0.02 |
| Ni | $< 0.02$ | 0.00–0.33 |
| Pb | 0.25 | 0.01–1.78 |
| Zn | 0.50 | 0.05–0.84 |

1–2 mg $l^{-1}$. However, long retention times in sewers should be avoided and, where this is not possible, then re-aeration, normally by oxygen injection, should be included in sewer design. Hydrogen sulphide should not be present in sewers if they are properly vented. However, if blockages occur or the flow falls to below 0.52 $ms^{-1}$ then anaerobic conditions can develop and hydrogen sulphide be given off (Sec. 1.3.1).

### 1.2.3. *Other wastewaters*

The other wastewaters rich in organic materials and so readily degraded biologically are the agricultural and food processing wastes. Of particular importance at present, and this varies according to agricultural practice and manufacturing processes, are wastes from intensive animal rearing, silage production, food processing, and the dairy industry.

*Animal wastes*

Specialisation and the development of new methods in agriculture has led to the intensification of animal rearing and a departure from the traditional farming practice of returning all wastes back to the land as fertilizer, which avoided pollution and treatment. The adoption of intensive farm practices can lead to enormous numbers of animals being kept in comparatively small areas. For example, Lynch and Poole (1979) cite an American situation where 35,000 cattle were kept in a feed lot of less than one square mile, whereas some farm animals such as pigs and poultry are now raised

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almost exclusively indoors in specially constructed units. Sewage is comparatively weak compared to most animal wastes which have very high $BOD_5$ concentrations (Table 1.24). The characteristics of animal wastes has been extensively reviewed by Evans *et al.* (1978; 1980). Most farm animals produce large quantities of waste each day compared to man, and in terms of mean population equivalents based on the $BOD_5$ where a man = 1.0, then a cow = 16.4, a horse = 11.2, a pig = 3.0, a sheep = 2.45, and a chicken = 0.014 (Gloyna 1971). Another problem is that the waste has a high solids content and thus, unlike sewage, is not a liquid but either a semi-liquid or semi-solid (Fig. 1.6). It is, therefore, difficult to handle or pump unless dewatered or diluted respectively. For many farmers, the limiting factor in the development of intensive animal rearing is the disposal of the increased amounts of animal waste. In Britain, the population of farm animals in 1978 was approximately 3 million cows, 9 million cattle, 7 million pigs and 130 million poultry. Their waste in terms of population equivalents was of the order of 30 million, 50 million, 17 million and 13 million respectively, which is almost twice the organic load produced by the human population in Britain (Weller and Willetts 1977). It is neither feasible nor desirable to discharge this quantity or type of waste to the public sewer. First, because of the vast volume of dilution water required to reduce the BOD to treatable levels by conventional wastewater treatment methods, and secondly because of the cost of increasing the treatment capacity of existing works by 200%. The strength of animal wastes compared to sewage and other agricultural wastewaters (Table 1.24) must be seen in relation to the dilution of the effluent. The daily volume of effluent produced by the major categories of animals being dairy cow 0.0445 $m^3$, beef cattle 0.0198 $m^3$, sow 0.0117 $m^3$, fat pig 0.0049 $m^3$, and poultry 0.0001 $m^3$ (Gowan 1972). A particular problem with animal wastes is the enhanced metal concentrations that are often present. Concentrated feeds used for fattening pigs in intensive rearing units contain high concentration of metals, especially copper and zinc. Research on a range of commercial pig foods has shown that these metals are present at concentrations ranging from 116–233 ppm dry weight for copper and 194–300 ppm for zinc. However, little of the metals is retained by the animals with between 70–80% of the copper and 92–96% of the zinc being excreted (Table 1.25). Effluents from intensively reared stock which are fed concentrates, and in particular pigs, will inevitably contain high concentrations of metals (Priem and Maton 1980), as well as other additives such as antibiotics.

Table 1.24. Average volume, strength, and nutrient content of animal wastes.

| Animal | Volume of waste per adult animal ($m^3$ $d^{-1}$) | COD (mg $l^{-1}$) | BOD (mg $l^{-1}$) | N (kg $tonne^{-1}$) | P (kg $tonne^{-1}$) | K (kg $tonne^{-1}$) | Moisture content (%) |
|---|---|---|---|---|---|---|---|
| Cow | 0.0500 | 150000 | 16100 | 11.1 | 4.5 | 13.4 | 87 |
| Pig | 0.0045 | 70000 | 30000 | 8.9 | 4.5 | 4.5 | 85 |
| Poultry | 0.0001 | 170000 | 24000 | 38.0 | 31.3 | 15.6 | 32–75[a] |

[a] Depending on housing.

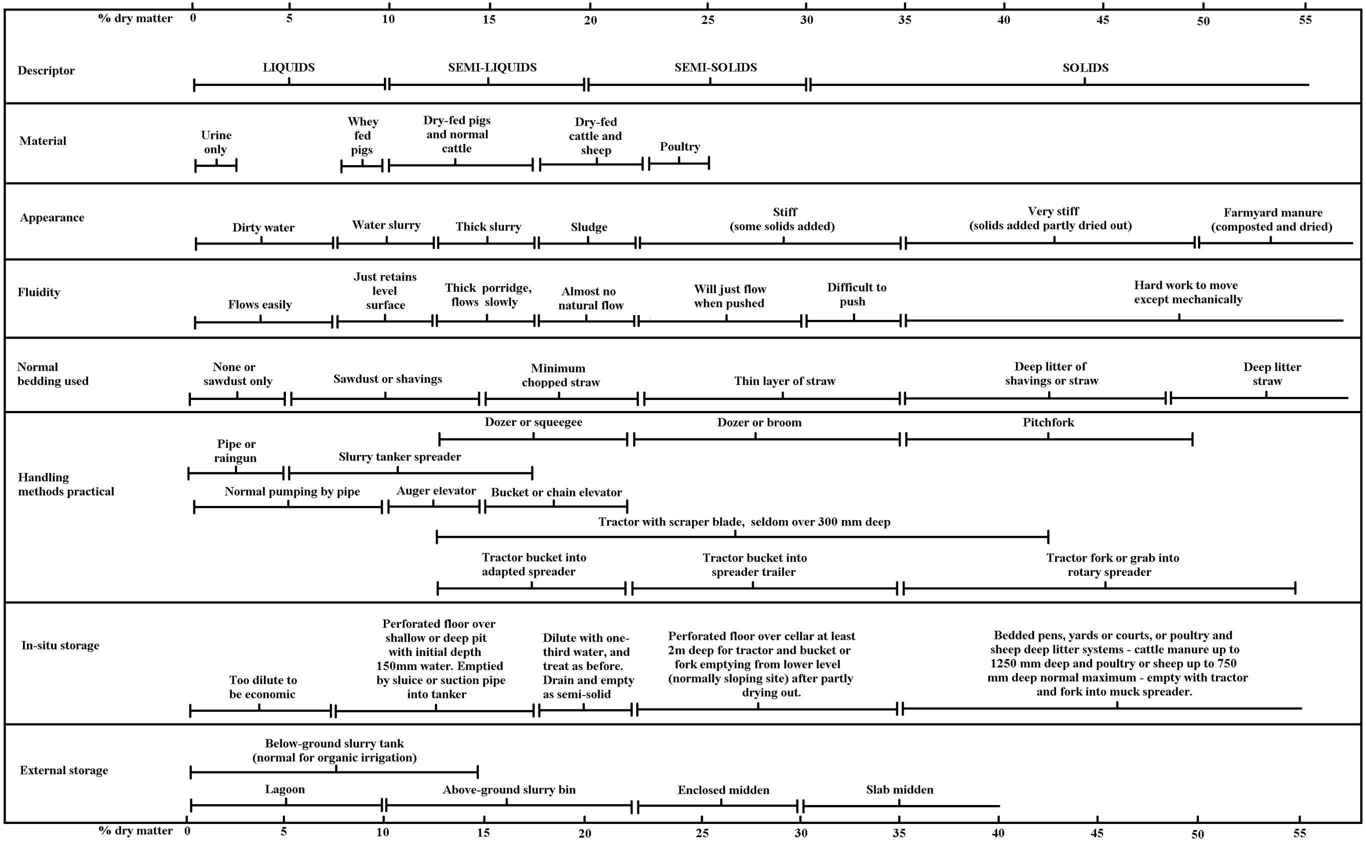

Fig. 1.6. Definition, handling, and storage of livestock effluent (Weller and Willets 1977).

Table 1.25. Average composition of the liquid manure from intensively reared pigs fed on commercial food concentrates rich in metals. Feed A, B and C contained 116, 233 and 189 ppm dry weight of copper and 194, 300 and 260 ppm dry weight of zinc respectively (adapted from Preim and Maton 1980).

| | A | | B | | C | |
|---|---|---|---|---|---|---|
| | $\bar{x}$ | S.D. | $\bar{x}$ | S. D. | $\bar{x}$ | S.D. |
| Dry matter (%) | 16.2 | 1.19 | 15.9 | 0.75 | 14.6 | 2.14 |
| Ash in dry matter (%) | 23.9 | 2.20 | 24.6 | 1.79 | 25.6 | 2.48 |
| BOD (mg $l^{-1}$) | 41807 | 1293 | 41967 | 2504 | 35546 | 5253 |
| COD (mg $l^{-1}$) | 163539 | 26051 | 154026 | 15867 | 143178 | 19997 |
| P (mg $l^{-1}$) | 3548 | 546 | 3491 | 360 | 3294 | 838 |
| Ammonia nitrogen (mg $l^{-1}$) | 6606 | 1343 | 6351 | 1097 | 5774 | 1209 |
| Kjeldhal nitrogen (mg $l^{-1}$) | 10345 | 1272 | 9998 | 1000 | 9083 | 1367 |
| Cu (ppm in dry matter) | 416 | 44.1 | 859 | 68.2 | 754 | 66.3 |
| Zn (ppm in dry matter) | 851 | 74.5 | 1385 | 107.4 | 1180 | 31.2 |
| As (ppm in dry matter) | 8.50 | 0.83 | 10.93 | 1.76 | 11.13 | 0.87 |
| Se (ppm in dry matter) | 0.63 | 0.05 | 1.57 | 0.17 | 1.08 | 0.08 |

### *Silage liquor*

Ensiling is a large-scale microbial process in which cut grass is degraded anaerobically so that the complex cellulose component is broken down into simpler organic acids, preserving the grass as food for cattle by raising the pH. The effluent (silage liquor) from the clamp in which the grass is stored is a mixture of surface water and plant juices from the ensiled herbage. It has a pH of 4.5 or less and is composed mainly of organic acids, in particular lactic, acetic, propionic, and butyric, which are very readily broken down biologically at a rate about four times faster than sewage, thus making it up to 1,000 times more potent as a pollutant (Patterson 1981). Thus, with an average BOD of 30,000–80,000 mg $l^{-1}$ (Beck 1989), discharges of the acidic liquor to watercourses lead to whole-sale destruction of all aerobic life. Silage liquor is also rich in nitrogen and can contain up to 2.5 g of nitrogen per litre of liquor (Weller and Willetts 1977). The dry matter content of silage liquor ranges from 4–10% (mean value of 6.5%). The average composition of the liquor as a percentage of the dry matter is crude protein 25.0%, ash 22.0%, lysine 1.0%, calcium 2.2%, phosphorus 1.0%, nitrogen-free extract 53.0%, lactic acid 25.0%, and volatile fatty acids 5.5% (Patterson 1980). The amount of liquor produced by silage is closely related to moisture content of the grass at the time of ensiling with the quantity of effluent per tonne of silage being 360–450 $l^{-1}$ at 10–15% dry matter, 90–225 l at 16–20%, and less than 90 l at dry matters of 25% and over (Gibbons 1968). The variation in the volume of effluent produced is also dependant on rainfall. Thus, the percentage of agricultural related water pollution incidents due to silage liquor varies from 14% in a dry year to 25% in a wet year (Haigh 1994). Between 1995 and 1998, there were on average 153 silage liquor related pollution incidents annually (range 114–234) in England and Wales (Environment Agency 1999). Acidic additives, used to preserve the silage, can increase the volume of liquor produced by up to 25%, although finely chopping silage does not have an effect on effluent production. While the volume of liquor can be greatly reduced by wilting the grass in the field before ensiling, occasionally wilting is not possible due to time or weather making the disposal of liquor a serious problem. It is not generally advisable to dispose of these effluents to the public sewer as small sewage works can be put out of action due to the resulting toxic shock. Such wastes are generally stored with the animal wastes in slurry tanks or stored in specially constructed acid-resistant tanks (O'Donnell *et al.* 1995a,b) and returned to the land when possible (Burford 1976). Some research has been undertaken on feeding fresh silage liquor to pigs (Patterson and Kilpatrick 1991), although this has not been widely adopted. Approximately $42.3 \times 10^6$ tonnes of forage crops were cut for silage in the UK during 1998, producing an estimated $2.12 \times 10^9$ litres of liquor (MAFF 1999).

### *Dairy industry*

Milk production has steadily grown over the past 30 years such that the dairy industry has now become the major agricultural processing industry in Europe.

Wastewater originates from two major processes, from the fluid milk itself at reception and bottling plants but more importantly at the processing plants that produce butter, cheese, evaporated and condensed milk, milk powder and other milk products. Milk itself has a $BOD_5$ of 100,000 mg $l^{-1}$ and washings from plants producing butter and cheese can have a $BOD_5$ ranging between 1,500–3,000 mg $l^{-1}$. Dairy wastes are dilutions of whole milk, separated milk, butter milk and whey. They are high in dissolved organic matter mainly in the form of the proteins (3.8%) casein and albumin, fat (3.6%) and lactose (4.5%) but low in suspended solids except for the fine curd found in cheese wastes. Nitrogen and phosphorus are also present, originating mainly from milk proteins (Guillen-Jimenez *et al.* 2000). Apart from whey, derived from the manufacture of cheese which is acidic, most dairy wastes are neutral or slightly alkaline but have a tendency to become acidic quite rapidly due to the fermentation of lactose to lactic acid. The average composition of milk, milk by-products and cheese wastes are given in Table 1.26. Details of the various processes used in the dairy industry, with specific reference to wastewater production, are given by Nemerow and Agardy (1998).

### *Food processing industries*

Waste from food processing is similar in nature to the food itself. Some processes give rise to large volumes of weakly polluted effluents such as vegetable washing water, which only contains soil and small amounts of organic matter. More concentrated wastewaters come from processes that either prepare the food or transform it in some way, such as the blanching of vegetables or pickling of meat. Generally, these wastes are rich in organic matter and normally contain sufficient nitrogen, phosphorus and trace elements for biological growth. The volume and strength of wastewater from food processing greatly depends on the type of process, the size and age of the plant as well as the season.

Cannery wastewaters are essentially the same as domestic kitchen waste. The waste originates from trimming, culling, juicing and blanching fruit and vegetables. The wastewaters are high in suspended solids, colloidal and dissolved organic matter, the main components being starch and fruit sugars. For example, 85–90% of the organic waste from a pineapple cannery is sugar in the form of sucrose (Painter 1971). Details of these wastes are summarised by Nemerow and Agardy (1998). Sugar beet waste is also comprised of sugars, 95% of which is sucrose with raffinose making up most of the remainder, although the waste is particularly low in nitrogen and phosphorus. The sugars are leached from cut and damaged surfaces into the transport and wash-water circuits, so that the BOD of these wastes can be as high as 8,000–10,000 mg $l^{-1}$ (Table 1.27). The accumulated sugars are rapidly catabolised in the circuits to short chain aliphatic carboxylic acids, principally acetic, propionic, and butyric acids, so that the wastewater requiring treatment is comprised almost exclusively of these acids. However, at low pH concentrations offensive odours from volatile fatty acids and sulphides can be generated, and sufficient lime (CaO) must be added to maintain circulating water at a neutral pH (Shore *et al.* 1984; Gray 1988). Brewery and distillery wastewaters are high in dissolved solids which contain nitrogen and fermented starches and their products. Fermentation wastes and in particular spent yeast is extremely concentrated with

Table 1.26. Composition and organic strength of milk products and associated waste products (Nemrow 1978).

| Characteristics | Whole milk (mg $l^{-1}$) | Skim milk (mg $l^{-1}$) | Buttermilk (mg $l^{-1}$) | Whey (mg $l^{-1}$) | Process wastes (mg $l^{-1}$) | Separated whey (mg $l^{-1}$) |
|---|---|---|---|---|---|---|
| Total solids | 125000 | 82300 | 77500 | 72000 | 4516 | 54772 |
| Organic solids | 117000 | 74500 | 68800 | 64000 | 2698 | 49612 |
| Ash solids | 8000 | 7800 | 8700 | 8000 | 1818 | 5160 |
| Fat | 36000 | 1000 | 5000 | 4000 | | |
| Soluble solids | | | | | 3956 | 54656 |
| Suspended solids | | | | | 560 | 116 |
| Milk sugar | 34000 | 46000 | 43000 | 44000 | | |
| Protein (casein) | 38000 | 39000 | 36000 | 8000 | | |
| Total organic nitrogen | | | | | 73.2 | 1300 |
| Free ammonia | | | | | 6.0 | 31 |
| Na | | | | | 807 | 648 |
| Ca | | | | | 112.5 | 350 |
| Mg | | | | | 25 | 78 |
| K | | | | | 116 | 1000 |
| P | | | | | 59 | 450 |
| $BOD_5$ | 102500 | 73000 | 64000 | 32000 | 1890 | 30100 |
| Oxygen consumed | 36750 | 32200 | 28600 | 25900 | | |

Table 1.27. Comparative strengths of wastewaters from food-processing industries.

| | BOD (mg $l^{-1}$) | COD (mg $l^{-1}$) | PV (mg $l^{-1}$) | Suspended solids (mg $l^{-1}$) | pH | Population equivalent per $m^3$ of waste |
|---|---|---|---|---|---|---|
| Brewery | 850 | 17000 | — | 90 | 4–6 | 14.2 |
| Cannery | | | | | | |
| citrus | 2000 | — | — | 7000 | Acid | 33.3 |
| pea | 570 | — | — | 130 | Acid | 9.5 |
| Dairy | 600–1000 | — | 150–250 | 200–400 | Acid | 10.0–16.7 |
| Distillery | 7000 | 10000 | — | Low | — | 116.7 |
| Farm | 1000–2000 | — | 500–1000 | 1500–3000 | 7.5–8.5 | 16.7–33.3 |
| Silage | 50000 | — | 12500 | Low | Acid | 833.3 |
| Potato processing | 2000 | 35000 | — | 2500 | 11–13 | 33.3 |
| Poultry | 500–800 | 600–1050 | — | 450–800 | 6.5–9.0 | 8.3–13.3 |
| Slaughterhouse | 1500–2500 | — | 200–400 | 800 | 7 | 25.0 |
| Sugar beet | 450–2000 | 600–3000 | — | 800–1500 | 7–8 | 7.5–33.3 |

the BOD (2,000–15,000 mg $l^{-1}$), total nitrogen (800–900 mg $l^{-1}$) and phosphate (20–140 mg $l^{-1}$), almost entirely present in the dissolved or colloidal fractions with the suspended solids content rarely in excess of 200 mg $l^{-1}$. Slaughterhouse and meat packing wastewaters are strong and unpleasant, being comprised of faeces and urine, blood washings from carcasses, floors and utensils, and the undigested food from the paunches of slaughtered animals. These wastewaters are high in dissolved and suspended organic matter, in particular proteins and fats, high in organic nitrogen and grease, as well as pathogens.

The strengths and volumes of wastewaters from the main food-processing industries are summarised in Table 1.27.

## 1.3. Micro-organisms and Pollution Control

Micro-organisms have a number of vital functions in pollution control. It is the microbial component of aquatic ecosystems that provides the self-purification capacity of natural waters in which micro-organisms respond to organic pollution by increased growth and metabolism (Sec. 1.4.1). It is essentially the same processes which occur in natural waters that are