Dispersion and self-purification of pollutants in surface water systems

A contribution to the International Hydrological Programme

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A contribution to the
International Hydrological
Programme

Dispersion and self-purification of pollutants in surface water systems

A report by IHP working group 6.1.

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UNESCO
Although the total amount of water on earth is generally assumed to have remained virtually constant, the rapid growth of population, together with the extension of irrigated agriculture and industrial development, are stressing the quantity and quality aspects of the natural system. Because of the increasing problems, man has begun to realize that he can no longer follow a "use and discard" philosophy—either with water resources or any other natural resource. As a result, the need for a consistent policy of rational management of water resources has become evident.

Rational water management, however, should be founded upon a thorough understanding of water availability and movement. Thus, as a contribution to the solution of the world's water problems, Unesco, in 1965, began the first world-wide programme of studies of the hydrological cycle—the International Hydrological Decade (IHD). The research programme was complemented by a major effort in the field of hydrological education and training. The activities undertaken during the Decade proved to be of great interest and value to Member States. By the end of that period, a majority of Unesco's Member States had formed IHD National Committees to carry out relevant national activities and to participate in regional and international co-operation within the IHD programme. The knowledge of the world's water resources had substantially improved. Hydrology became widely recognized as an independent professional option and facilities for the training of hydrologists had been developed.

Conscious of the need to expand upon the efforts initiated during the International Hydrological Decade and, following the recommendations of Member States, Unesco, in 1975, launched a new long-term intergovernmental programme, the International Hydrological Programme (IHP), to follow the Decade.

Although the IHP is basically a scientific and educational programme, Unesco has been aware from the beginning of a need to direct its activities toward the practical solutions of the world's very real water resources problems. Accordingly, and in line with the recommendations of the 1977 United Nations Water Conference, the objectives of the International Hydrological Programme have been gradually expanded in order to cover not only hydrological processes considered in interrelationship with the environment and human activities, but also the scientific aspects of multi-purpose utilization and conservation of water resources to meet the needs of economic and social development. Thus, while maintaining IHP's scientific concept, the objectives have shifted perceptibly towards a multidisciplinary approach to the assessment, planning, and rational management of water resources.

As part of Unesco's contribution to the objectives of the IHP, two publication series are issued: "Studies and Reports in Hydrology" and "Technical Papers in Hydrology". In addition to these publications, and in order to expedite exchange of information in the areas in which it is most needed, works of a preliminary nature are issued in the form of Technical Documents.

The "Technical Papers in Hydrology" series, to which this volume belongs, is intended to provide a means for the exchange of information on hydrological techniques and for the coordination of research and data collection. Unesco uses this series as a means of bringing together and making known the experience accumulated by hydrologists throughout the world.
## Contents

### FOREWORD

1. GENERAL SCIENTIFIC BACKGROUND
   1.1 Definitions of pollution
   1.2 Processes controlling pollution behaviour
   1.3 Measurement of pollutants

2. POLLUTANTS AND SOURCES OF POLLUTION
   2.1 Water pollutants
   2.2 Sources of pollution
      2.2.1 Community waste waters
      2.2.2 Industrial waste waters
      2.2.3 Pollutants from agriculture
      2.2.4 Other potential sources of pollution

3. CHARACTERISTICS OF RIVER SYSTEMS
   3.1 Introduction
   3.2 Physical and hydraulic behaviour
      3.2.1 Water velocity and streamflow regimes
      3.2.2 Dispersion of pollutants
      3.2.3 Mass balance equations
   3.3 Self purification and the oxygen balance
      3.3.1 Biochemical oxygen demand
      3.3.2 Immediate chemical oxygen demand
      3.3.3 Plant respiration and photosynthesis
      3.3.4 Atmospheric reaeration
   3.4 Prediction of water quality conditions
      3.4.1 Longitudinal spreading of a conservative pollutant
      3.4.2 Longitudinal spreading of a non-conservative pollutant
      3.4.3 Oxygen balance
      3.4.4 Management applications of flow and water quality models
   3.5 Major types of biotic changes caused by selected pollutants
      3.5.1 Introduction
      3.5.2 Substances which settle or cause turbidity
      3.5.3 Oxygen consuming substances
      3.5.4 Stages of biotic change

4. CHARACTERISTICS OF LAKES AND RESERVOIRS
   4.1 Introduction
   4.2 Physical and hydraulic characteristics
      4.2.1 Thermal stratification
      4.2.2 Water movement
   4.3 Chemical and biological characteristics
   4.4 Dispersion of pollutants
      4.4.1 Diffusion processes
   4.5 Application of hydrodynamic considerations to assessing pollution problems
   4.6 Self-purification processes
<table>
<thead>
<tr>
<th>Section</th>
<th>Title</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>4.6.1</td>
<td>Introduction</td>
<td>90</td>
</tr>
<tr>
<td>4.6.2</td>
<td>Oxidation</td>
<td>90</td>
</tr>
<tr>
<td>4.6.3</td>
<td>Biological activity</td>
<td>92</td>
</tr>
<tr>
<td>4.6.4</td>
<td>Sedimentation</td>
<td>93</td>
</tr>
<tr>
<td>4.7</td>
<td>Major kinds of biotic change caused by selected pollutants</td>
<td>93</td>
</tr>
<tr>
<td>4.7.1</td>
<td>Substances causing eutrophication</td>
<td>93</td>
</tr>
<tr>
<td>4.7.2</td>
<td>Inhibiting or toxic substances</td>
<td>95</td>
</tr>
</tbody>
</table>
Foreword

The price paid for industrial civilization is the ever-increasing quantity of waste materials requiring safe disposal. Studies therefore need to be made of the manner in which waste products are accumulated, dispersed, or destroyed in the environment, with the object of understanding the relationships involved so that regulations can be devised to safeguard the essential characteristics of that environment. Surface water in particular, because of its mobility, its high replacement rate in the hydrological cycle and its relative abundance in populated regions, is an excellent vehicle for the discharge and dispersion of many types of waste products in rivers, lakes and reservoirs. It is important, therefore, to develop methods for evaluating the capacity of the different environments for the reception of liquid effluents and/or solid wastes without significant damage occurring.

A first approach was made during the International Hydrological Decade (IHD) which led to the Unesco/WHO publication 'Water Quality Surveys' (1978). The pollution of water became an important issue for the International Hydrological Programme (IHP). IHP Project 6 covers a number of scientific problems related to water quality and to water pollution.

The Intergovernmental Council of the IHP established four sub-projects on the quality of water. The present report is the response to the Council's resolution on project 6.1 'Study of diffusion, dispersion and self-purification processes of pollutants in rivers, lakes, reservoirs and estuaries'.

When introducing project 6.1 the Council stated (Report on the First Session of the Intergovernmental Council of the IHP, Unesco, 1975):

'The choice of the best methods of defining and monitoring the effects of human activity on surface-water quality depends on the conditions pertaining at the site under consideration (information available, characteristics of the aquatic environment into which pollutants are discharged, types of pollution, etc.). It is essential also to understand the physical, chemical and biological processes of dispersion and self-purification which are at work in the different types of surface water.

Progress in industrialized countries has enabled man to acquire much greater knowledge of the mechanics of the movement and dispersion of pollutants in aquatic environments and of self-purification processes; it is important, therefore, to take stock of this knowledge in order to provide a sound basis for the task of developing tools for evaluating the acceptance capacity of rivers, lakes and reservoirs.'

To reach this objective, the Intergovernmental Council of the IHP decided that a state-of-the-art report should be prepared on:

1. The mechanisms of the movement and dispersion of pollutants in rivers, lakes, reservoirs and estuaries; and
2. The self-purification processes taking place in the above water bodies.

To this end, the Council established a working group with Messrs. R. L. Casañas (Argentina), G. Friedrich (Federal Republic of Germany), K. Ivanov (Bulgaria) and A. S. Lobato de Faria (Portugal; Chairman) as members.
The main aim of this publication is to provide hydrologists and engineers involved with water quality problems with a 'state-of-the-art' report on the processes which control the distribution and behaviour of pollutants in aquatic media and on the major types of biotic changes caused by some of them. It is hoped that the work will provide the scientific basis for decisions concerning the study of water quality problems and of the factors to be taken into account in surveillance programmes and other activities.

Following the foreword a general scientific background is given in Chapter 1 which contains some discussion on the scope and limitations of this publication.

Chapter 2 gives some characteristics of sources of pollution and different types of pollutants. It distinguishes conservative wastes from non-conservative wastes, and presents some characteristics of the receiving media.

Chapter 3 describes physical, chemical and biological aspects of processes affecting water quality in river systems and Chapter 4 considers processes affecting lake and reservoir systems.

All chapters have been discussed in some detail within the IHP working group on project 6.1. Textual revision and final editing were performed by the chief editors, Dr P. G. Whitehead, Institute of Hydrology, Wallingford, Oxon, UK and Dr T. Lack, Water Research Centre, Medmenham, Bucks, UK.
1 General scientific background
1.1 DEFINITIONS OF POLLUTION

Although this publication makes a state-of-the-art approach to the mechanisms of dispersion and self-purification, this does not mean that it provides the reader with a manual on the evaluation of the 'acceptance capacity' of fresh surface water bodies for any arbitrary pollutant. The reason for this is the complexity of pollution processes especially when coupled with the institutional arrangements which surround any practical case of pollution. A universally applicable and acceptable definition of pollution and pollutants is therefore inconceivable and in the light of this it is not surprising that no satisfactory definition of the acceptance capacity can be given. This may be substantiated by citing some authoritative opinions and statements published during the past 30 years or so.

Coulson and Forbes (1952), writing about rivers and the common law, state that pollution is 'the addition of something to water which changes its natural qualities so that the riparian proprietor does not get the natural water of the stream transmitted to him'. This concept is far reaching as the authors themselves stress and includes as polluting acts such processes as changing hard water to soft water, the raising of the temperature of water, and the addition of an agent which, on meeting some other substances already in the water, each in themselves being harmless, causes pollution.

Klein (1962) also refers to the very wide significance of the common law approach, and states: 'From the scientific standpoint, it is perhaps easier to regard pollution as the actual impurity introduced into the stream rather than the act of introducing such impurity, and to define pollution as anything causing or inducing objectionable conditions in any water course and affecting adversely any use or uses to which the water thereof may be put'.

As can be seen, this is a definition of both 'pollutant' and 'pollution', the latter being derived from the effects caused or induced by the former.

It is perhaps of interest to introduce here a distinction between 'contaminant' and 'pollutant', which in some parts of the world have different connotations. The State of California, for example, in a law of 1950, defines contamination as the 'impairment of water quality by sewage or industrial waste causing an actual hazard to public health or an equivalent effect', pollution being regarded as anything which 'adversely and unreasonably impairs the beneficial use of water even though actual health hazard is not involved'. The well-known textbook by Fair et al. (1966) follows a similar approach. In some European and African countries, 'contamination' of water is linked with biological agents which play a role in transmissible diseases, the word 'pollution' being connected with physical agents or chemical substances which cause health hazards or impair the uses of water.

Another important definition of water pollution was included in the broad statement given by the National Academy of Sciences, USA, in which 'pollution' is an undesirable change in the physical, chemical or biological characteristics of our air, land and water that may or will harmfully affect human life, the lives of desirable species, our industrial processes, living conditions and cultural assets; or that may or will waste or deteriorate our raw material resources; pollutants are residues of the things we may use or throw away'. Commenting on this, Odum (1971) notes that 'throw-away' pollutants must include pollutants that are released as by-products of transportation, industry and agriculture.

In the far-reaching book 'Health Hazards of the Human Environment', the World Health Organization (1972) made their first systematic definition of water pollution: 'water is considered polluted when it is altered in composition or condition so that it becomes less suitable for any or all of the functions and purposes for which it would be suitable in its
At about the same time, water control regulations promulgated in the USSR, state in article 2, that surface waters must be considered as polluted 'when their properties have been modified, under the direct or indirect influence of an industrial activity and/or the life conditions of the population, up to a degree which make them unsuitable for any use or uses for which they are intended' (USSR, 1971).

This approach followed the definition given by Alekhin in 1970, that by 'pollution one should understand any deviation of water properties (chemical, physical or biological) from their natural condition, together with any deterioration of quality from the standpoint of the utilisation of water for practical purposes'.

In an attempt to meet the aims of the different approaches, the following definition of water pollution is offered: any modification either natural or artificial, which, directly or indirectly, changes the quality of water and disturbs or destroys the balance of ecosystems and natural resources, so that it (1) causes hazards to public health, (2) detracts from the convenience, efficiency and well-being of man and his communities and (3) impairs the beneficial uses of water.

1.2 PROCESSES CONTROLLING POLLUTANT BEHAVIOUR

In order to assess the impact of a pollutant on the aquatic environment quantitative information is required about
- the nature of the waste being discharged,
- the quantity of waste being discharged,
- the capacity of the receiving medium to accept the pollutant.

For a given input, the concentrations found in the receiving medium are affected by the dilution and dispersion occurring within the problem area and the reactions occurring between pollutants and chemical and biological interactions with the surrounding environment during transportation. As far as the physical factors determining the fate of pollutants is concerned, two principal hydrodynamic processes can be distinguished:
1. Advective transport (associated with a net displacement of a body of water);
2. Diffusive or dispersive transport (associated with mixing processes within the body of water).

To determine the importance of the advective transport process the velocity field must be known; and to determine the extent of dispersion processes the effect of turbulence must be characterized. As the velocity field and the degree of turbulence vary with the nature of the water body under investigation a distinction has to be made between rivers, lakes and reservoirs because each possesses different hydrodynamic characteristics. These processes are considered in detail for different types of water bodies in later chapters of this report.

In addition to advective and dispersive effects the processes of sedimentation and resuspension are also important in determining the physical redistribution of pollutants. For example, organic matter downstream of an effluent discharge may settle on the river bed under low flow conditions only to be resuspended with a sudden increase in flow. These processes are particularly important in evaluating pollutants associated with sediments. The pollution problems associated with sediments are not considered in detail in this report since they are reported elsewhere (Golterman, 1981).

The extent of chemical reactions on pollutant behaviour varies with the nature of the pollutant. Typical sources of pollution, namely community waste waters, industrial waste waters, agricultural waste waters (eg fertiliser runoff) and others release a wide variety of substances.

Pollutants that enter water courses as a result of man's domestic, industrial and agricultural activities can be grouped in a number of ways. One very broad division, which emphasises changes within the river system, distinguishes between conservative and non-conservative water quality variables.

With conservative variables the mass of the pollutant is not altered by the chemical and biological processes occurring in surface waters. For the most part these are the inorganic constituents such as chlorides which, once they enter the river, are diluted and dispersed but not appreciably changed in quantity.

By contrast, non-conservative variables undergo some form of chemical or biological change. The reactions determining such change are often complex. They may involve the interaction between water quality variables and environmental factors. For example the distribution of oxygen in a stream is a balance between various sources and sinks of oxygen. On the one hand there is the oxygen supplied by re-aeration from the atmosphere and photosynthetic oxygen produced by plants and algae and, on the other hand, oxygen is being consumed by respiration processes and the removal of oxygen during the biochemical breakdown of organic material from
effluent. There are, of course, many other factors that can influence the oxygen regime in a river and these will be discussed in later chapters. However, it is important to recognize that a balance exists between deteriorating water quality conditions and the natural recovery processes and in order to assess the effect of a pollutant it is necessary to understand the processes controlling this balance.

With regard to the effects of pollution, two kinds of pollutants can be distinguished:
1. pollutants having an immediate and 'once and for all' effect;
2. pollutants whose effects are cumulative over a period of time.

An immediate increase in the BOD (Biochemical Oxygen Demand) content of river waters caused by the discharge of organic material in effluents is an example of type (1). Examples of pollutants whose effects persist or occur as a result of accumulation are:
1. the accumulation of mercury in sediments; the transport of the contaminated sediments and subsequent deposition in places remote from the source, followed by the remobilization of the mercury at the site of deposition;
2. the discharge of potentially dangerous constituents at less than acutely toxic concentrations followed by their accumulation in the food chain (algae, fish, animals, man) to such an extent that the eating of fish by man, for example, would have to be limited;
3. the release of nutrients leading to eutrophication which may change fish populations in such a way that high quality fish may be eliminated and replaced by less desirable species. The question of eutrophication is considered later in this report.

The complexities inherent in the behaviour of surface waters, coupled with the need for quantitative descriptions of water quality, have created great interest in water quality models as tools for simulating the response of water quality variables to alternative basin planning proposals. The study of dissolved oxygen (DO) - biochemical oxygen demand (BOD) relationships is generally considered to be a fundamental feature of applied water quality modelling. Although DO continues to be the subject of a majority of water quality models, other variables and processes are receiving increasing attention. The subjects range in complexity from relatively simple variables, such as temperature, to highly complex, long-term processes such as eutrophication. The question of modelling water quality is considered in later chapters of the report and the reader is referred to the wide spectrum of literature in this area (Thomann, 1972; IAHS Baden Symposium, 1978).

1.3 MEASUREMENT OF POLLUTANTS

Quantitative and qualitative measurements are indispensable if the processes affecting the pollution of fresh surface waters are to be understood and controlled.

The recent Unesco/WHO publication on Water Quality Surveys (Unesco/WHO, 1978) devotes a short sub-chapter (pages 24-28) to the objectives and planning of networks for water quality measurements. Because the scope of the present report is contained within the larger objective of water quality control, the reader is referred to the above book for further information.

For present purposes, the definitions of the terms 'monitoring', 'survey' and 'surveillance', which are contained in the Unesco/WHO publication, have been adopted. These are:

Monitoring: the continuous, standardized measurement and observation of the environment;

Survey: a series of intensive programmes, of finite duration, designed to measure and observe the environment in detail for a specific purpose;

Surveillance: the continuous specific observation and measurement of the environment relative to control or management.

All three types of programme are important for an analysis of the dispersion of pollutants and self-purification processes occurring in rivers, lakes and reservoirs, and have the following aims:
1. to provide a basic set of data for the purpose of developing scientific knowledge, including water quality modelling;
2. to protect a body of water against pollution, and to make it available for beneficial use elsewhere;
3. to ensure the quality of water for different uses (drinking water supply, irrigation, watering of livestock, fish culture, recreation, etc.);
4. to forecast the quality of water in order to ensure that water quality standards are not violated;
5. to help solve acute problems caused by pollution.

All three types of measurement programmes may be suitable for the purposes of analysing dispersion and self-purification processes. The choice of a particular programme depends on the nature of the pollution problem.
Broadly speaking, a monitoring programme would be desirable in order to define the characteristics of the water resource, the nature of existing inputs and their potential pollutant constituents in such a way that it would be easy to devise a management strategy to control pollution at any time.

However, it is impossible, for practical reasons related to both economic and human resources, to have an all-embracing monitoring programme which covers every situation arising in practical problems of pollution. For this reason surveillance programmes are considered to be the most important in the field of pollution control.

Intensive survey programmes are important because they determine the acceptance capacity of the receiving media under the conditions of the survey, and provide additional information to enable a model to be developed to simulate the behaviour of the system under investigation.

When carrying out a measurement programme the question arises as to whether sampling should take place in the polluting effluent or the receiving waters. Sampling of the effluents is easier, because the pollutants are normally present at their maximum concentrations. This type of measurement would be essential in environmental protection if standards were to be enforced. Sampling in the receiving waters is essential for providing data for environmental modelling, and developing a strategy for water resource management. Most programmes will require careful sampling in both media.

1.3.1 Water quality variables to be measured

The discussion of which determinands are suitable for measurement is outside the scope of this report, and the reader is referred to the report of a Workshop held in Reading, United Kingdom, in January 1977 (WHO, 1977), on the optimization of water quality monitoring networks.

Table 1.1, taken from this publication, presents a comprehensive list of determinands related to water uses, water sources and sources of pollution and affords some guidance as to their relative importance in pollution control.

Whichever programme and determinands are selected, a few important factors have to be considered during the preparation of a water quality investigation. A detailed discussion of these can be found in the literature (Unesco/WHO, 1978). For present purposes it is enough for the reader to be aware of the important factors which are relevant to all measurement programmes. The overall objectives of the study and the specific objectives of the sampling programme must be clearly defined at the outset. Measurements of water quality are usually complex and expensive and it is important to be aware of the need for and the eventual use of these data. In addition to chemical and other forms of analysis, the physical behaviour of the water body need to be considered since there will have a major influence on the location of sampling points and the frequency of sampling.

Similar remarks pertain to the sampling of polluting effluents. Both flow and quality have to be properly measured if reliable results are to be obtained. In many countries hydrometric networks were established long before quality surveillance was begun. It does not follow that points selected for the measurement of water discharges or levels are the best for the purposes of water quality control planning and surveillance. A good example of this may be found at the junction of a small, highly polluted tributary with a large, cleaner river. For gauging purposes, a measuring station may be located close to the confluence, in the middle of the major river's cross-section, while for quality monitoring this point is unlikely to be the best location for sampling because it is unlikely that complete mixing will have occurred.

Finally, it should be remembered that future developments which change the hydraulic regime of the water body will almost certainly call for additional sampling to check the effect of the development. This should be anticipated by careful evaluation of the potential environmental impact.

In addition to the difficulty of designing an effective sample strategy, further uncertainty is introduced by the measurement errors associated with laboratory analysis. For example, Table 1.2 indicates the level of measurement errors obtained from a series of duplicated experiments for a range of water quality variables (Whitehead et al., 1981). Certain determinands (such as BOD) contain high levels of error compared to other determinands (such as chloride). It is important therefore to recognize the existence of such errors when planning a sampling programme and particularly when analysing the results and applying modelling techniques to aid management decisions.
Table 1.1 Determinands related to water uses, water sources and sources of pollution

- **Determinands of particular importance**
- **Determinands less significant but worth monitoring**

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* Determinands usually monitored for operational purposes
Table 1.1 continued

- Determinands of particular importance
- Determinands less significant but worth monitoring

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<tr>
<th>Group</th>
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**Inorganic II**

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*Determinands of particular importance*  
*Determinands less significant but worth monitoring*

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**Organic**

*Total monohydric phenols*  
Individual phenols  
Anionic detergents  
Cationic detergents  
Nonionic detergents  
*Hydrocarbon oils*  
-persistent  
-non-persistent  
-individual oils  
Organochlorine compounds  
-Individual Pesticides  
-Chlorine  
-Organophosphorus  
-Other  
PAH  
Organic acids  
Lignins  
Humic acids

**Radiochemical**

Gross α activity  
Gross β activity  
Radionuclides  
Individual γ emitters
Table 1.1 continued

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Δ Determinands of particular importance
☐ Determinands less significant but worth monitoring
Table 1.1 continued

\* Determinands of particular importance
\[ Determinands less significant but worth monitoring

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Olefactometrical

Odour

Chemical

General

- pH
- Conductivity
- Dissolved solids
- Ash
- Alkalinity
- Acidity
- Hardness
- BOD
- BOD (inhibited)
- TOD
- TOC
- COD
- Permanganate value
- UV-absorption

Gases

\( O_2 \)
\( CO_2 \)
\( Cl_2 \)

Chemical

Inorganic

- Ca
- Mg
- Na
- K
- Fe
- Mn
- Al
- HCO\(_3\)
- SO\(_4\)
- Cl
- SiO\(_2\)
- NO\(_3\)
- NO\(_2\)
- NH\(_4\)
- Organic N
- Total N
- Orthophosphate-P
- Total P
- Total phosphate-P

* Depending on disinfection treatment
Table 1.1 continued

- Determinands of particular importance
- Determinands less significant but worth monitoring

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Chemical Organic

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Anionic detergents | | | |
Cationic detergents | | | |
Nonionic detergents | | | |
Determinants of particular importance

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Δ Determinands of particular importance
□ Determinands less significant but worth monitoring
Table 1.2. Laboratory analysis errors based on 52 duplicate samples taken from the Bedford Ouse River in 1974 (Whitehead et al, 1981)

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* % error = standard deviation x 100/mean.

REFERENCES


California State and Regional Water Pollution Control Boards, 1952. Water Pollution Control - Progress Report - 1950 to 1952. WPCB Publ. No. 5, San Francisco.


2 Pollutants
and sources of pollution
Before describing the various processes which affect the quality of fresh surface waters, it is necessary to describe the nature of the substances which can be considered to be pollutants and routes by which they may enter a water body.

This chapter begins with a general discussion on water pollutants and then examines three well-defined sources of pollution: municipal, industrial and agricultural waste-waters.

Special importance is given to the classification of pollutants as conservative and non-conservative (the latter including bio-degradable substances) and to the distinction between point and diffuse sources.

2.1 WATER POLLUTANTS

Important aspects to be considered when describing the nature, source and mode of action of a pollutant are the health and ecological hazards along with the impairment of the beneficial uses of water. Problems caused by natural phenomena and those arising from the rejection of waste heat are not considered in this report. The effects of thermal discharges is considered in the Unesco report 'Predicting Effects of Power Plant Once-Through Cooling on Aquatic Systems', 1979.

There are several classifications of pollutants in the literature. Chanlett (1973) considered pollutants in relation to their objectionable characteristics. Although the original classification was concerned with industrial waste-waters, it is also applicable to domestic and agricultural discharges and these have been included in Table 2.1.

Table 2.1. Objectionable component groups of waste-waters, their effects and typical sources (after Chanlett, 1973)

<table>
<thead>
<tr>
<th>Component group</th>
<th>Effects</th>
<th>Typical sources</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Bio-oxidizable substances expressed as BOD₅</td>
<td>Deoxygenation, fish kills, production of odours</td>
<td>Sugar refineries; canneries; distilleries; breweries; milk processing; pulping and paper making and domestic sewage.</td>
</tr>
<tr>
<td>2. Poisonous substances: As, CN, Cr, Cd, Cu, Hg, Pb, Zn</td>
<td>Fish kills, livestock poisoning, plankton kills, accumulation in tissues of molluscs and fish</td>
<td>Metal cleaning, plating and pickling; phosphate and bauxite refining; chlorine production; battery making and tanning.</td>
</tr>
<tr>
<td>3. Acids and alkalis</td>
<td>Disruption of buffering capacity and disturbance to ecological systems</td>
<td>Coal mine drainage; steel pickling; textile manufacture; chemical manufacture; wool scouring and laundries.</td>
</tr>
</tbody>
</table>
Table 2.1. (continued)

<table>
<thead>
<tr>
<th>Component group</th>
<th>Effects</th>
<th>Typical sources</th>
</tr>
</thead>
<tbody>
<tr>
<td>Disinfectants: Cl₂, H₂O₂, formalin, phenol</td>
<td>Selective kills of micro-organisms. Production of tastes and odours in treated water</td>
<td>Paper and textile bleaching; resin synthesis; antibiotics preparation; gas, coke and coal-tar making; dye and chemical manufacture.</td>
</tr>
<tr>
<td>Salts: Fe, Ca, Mg, Mn, Cl, SO₄</td>
<td>Changed water characteristics such as hardness and salinity, scale formation and staining</td>
<td>Metallurgical processes; cement making; ceramics; oil well discharge; mine discharges.</td>
</tr>
<tr>
<td>Oxidising and reducing agents: NH₃, NO₂, NO₃, S, SO₃</td>
<td>Oxygen depletion, possible eutrophication, selective microbial growths, odours.</td>
<td>Gas and coke making; fertilizer making; explosive, dye and synthetic fibre manufacture; wood pulping and bleaching; domestic waste-water.</td>
</tr>
<tr>
<td>Aesthetically objectionable substances</td>
<td>Floating and settleable solids, eutrophication, anaerobic bottom deposits, odours, foaming, waterfowl and fish mortality</td>
<td>Detergent manufacture; tanning; food processing; sugar mills; woollen mills; poultry dressing; petroleum refining; oil spills; domestic waste-water.</td>
</tr>
<tr>
<td>Pathogenic organisms</td>
<td>Infections in man, reinfection of livestock, plant diseases through contaminated irrigation water</td>
<td>Abattoir wastes; wool processing; poultry processing wastes; fungal growth in waste treatment works.</td>
</tr>
</tbody>
</table>

A different approach is to classify pollution on the basis of its source and the factors affecting water quality deterioration, Table 2.2.

Table 2.2. Some common sources and causes of water quality deterioration (after Farrimond, 1980)

<table>
<thead>
<tr>
<th>Effluent source</th>
<th>Factors affecting water quality deterioration</th>
</tr>
</thead>
<tbody>
<tr>
<td>Domestic sewage</td>
<td>BOD, suspended solids, ammonia, nitrate, phosphate.</td>
</tr>
<tr>
<td>Vegetable processing</td>
<td>BOD, suspended solids, colour.</td>
</tr>
<tr>
<td>Chemical industry</td>
<td>BOD, ammonia, phenols, non-biodegradable organics</td>
</tr>
<tr>
<td>Iron and steel manufacture</td>
<td>Cyanide, phenols, thiocyanate, pH, ammonia, sulphides.</td>
</tr>
<tr>
<td>Coal mining</td>
<td>Suspended solids, iron, pH, dissolved metals.</td>
</tr>
<tr>
<td>Metal finishing</td>
<td>Cyanide, copper, cadmium, nickel, pH</td>
</tr>
<tr>
<td>Brewing</td>
<td>Suspended solids, BOD, pH.</td>
</tr>
<tr>
<td>Dairy products</td>
<td>BOD, pH.</td>
</tr>
<tr>
<td>Oil refineries</td>
<td>Ammonia, phenols, oil, sulphide.</td>
</tr>
<tr>
<td>Quarrying</td>
<td>Suspended solids, oil.</td>
</tr>
<tr>
<td>Agriculture</td>
<td>Nitrate, phosphate, BOD</td>
</tr>
</tbody>
</table>
Extensive lists of common pollutants and views as to their importance in reducing water quality have been produced by many government agencies. These can be summarised as follows:

1. Suspended particulate matter
2. Acids and alkalis
3. Plant nutrients (nitrates and phosphates)
4. Cyanide
5. Sulphite
6. Phosphorus
7. Metals and metalloids
8. Organic compounds (petrochemicals, plastics)
9. Organic wastes
10. Pesticides
11. PCBs (polychlorinated biphenyls)
12. Miscellaneous metal-based compounds
13. Radioactive nuclides
14. Detergents
15. Micro-organisms

Classifications, such as these, refer only to the deleterious effects to be expected on the basis of source and composition and do not provide much insight into the complex processes which occur following the discharge of a pollutant to a water-course.

To further an understanding of the processes, particularly from an ecosystem viewpoint (Odum, 1971), it is necessary to distinguish between non-degradable and biodegradable pollutants. To take account of non-biologically controlled processes it is also helpful to describe substances as being either conservative or non-conservative. For example, some non-degradable compounds can effectively be immobilized by sorption onto sediments in which case their behaviour would be described as non-conservative, i.e. their mass flux would decrease in passage downstream. Conservative substances are those which either do not change in character or chemical state with time or change imperceptibly slowly in relation to their retention time within a given system. Dilution is often the only economic way of abating pollution caused by these materials because removal processes are usually energy consumptive and therefore expensive (e.g. desalination). Recycling schemes, better process control and more efficient production technology are ways of minimizing the quantities of conservative substances released in waste-waters. Non-conservative substances are those which can rapidly be oxidized, immobilized or decomposed by natural processes either in the receiving stream or at the waste treatment works. Examples are: easily biodegradable organic substances contained in domestic sewage and food industry waste-water; settleable solids and chemicals which react rapidly after being introduced into receiving waters. Odum (1971) shows, in a simple figure (Figure 2.1), the effects of two kinds of pollutants: degradable organic and non-degradable toxic, on the energy available for production in ecosystems. Up to a certain level, degradable pollutants that provide energy or nutrients will increase the productivity of the ecosystem. This process has been called eutrophication. Higher levels of input over a critical range results in severe oscillations in the productivity. Additional input above this critical range becomes a stress and productivity is decreased. As shown in the lower part of Figure 2.1, toxic materials are stressful from the beginning and increasingly depress production as their concentration increases. However, this effect may be difficult to detect at low or sub-lethal levels.

Having discussed in broad terms the kinds of pollutants with which this report is concerned, it is now necessary to consider relationships between sources of pollutants and their effects. To do this, the most important effects of pollutants are grouped under six headings roughly following the system used by McGauhey (1968).

1. Physiological damage to man and useful organisms.
2. Transmission and release of pathogenic micro-organisms.
3. Reduction of the oxygen resources of water bodies caused by the biochemical oxidation of degradable substances and by a reduction in the expected rate of aeration through the surface.
4. Eutrophication leading to excessive growth of algae and higher plants and the formation of anaerobic waters and sediments.
5. Damage to food chains caused by the accumulation of poisonous substances in living organisms.
6. Impairment or destruction of aesthetic values.

Following a discussion of the sources of pollution, these groups are considered again in Table 2.3 and more detailed remarks about the effects of selected pollutants are given in Chapters 3 and 4.
Figure 2.1 The effects of two types of pollutants on the energy available for production. Upper diagram: organic degradable substance. Lower diagram: toxic non-degradable substance (from Odum, 1971)
2.2 SOURCES OF POLLUTION

Changes in the quality of inland, fresh, surface waters may be brought about by natural or artificial causes. Artificial additions which affect quality may be accidental or unintentional (as in the case of diffuse drainage) or deliberate (as in the case of authorized discharges). Since it is impossible to give a detailed discussion of all causes of pollution, this analysis is largely confined to authorizeable discharges which are usually the more obvious and important sources of pollution. (The reader will find a detailed discussion of accidental pollution of inland waters in the WHO publication EURO 3105 W (1972)).

Non-accidental sources may be classified by the ways in which pollutants are introduced into the aquatic environment and by the activity or process which generates the discharge. Effective control of pollution requires a clear understanding of the differences between diffuse and point sources of pollutants.

Point sources are discharges whose total flow is conveyed in a well-defined channel. Typical examples are municipal and industrial discharges of waste-water, storm water culverts and sewer overflows. These all possess the property that the total load of pollutants can be determined by sampling and flow measurement at the point of entry to a receiving watercourse.

Diffuse or non-point sources are laterally extended discharges where the total flow cannot be measured or sampled directly or even readily observed at a single point. The processes which pollute these discharges may be discrete or dispersively spread over the catchment. Thus the essential distinction between point and diffuse discharges (irrespective of the mechanism of pollution) is the ability to remove the pollutant from the former. This contrasts with the methods of control available for diffuse sources which can only be regulated by curtailing an activity, which if uncontrolled, would give rise to pollution, e.g. careless spraying of crops with pesticide or an excessive use of fertilizers on agricultural land. Because diffuse pollution often arises as a result of wholly desirable activities, the control mechanisms may cause economic and political problems which may not immediately be obvious and which require a fuller evaluation than that which might be given to controlling point sources of pollution.

With these definitions the sources of pollution can be classified as follows:

A. Sources capable of control at a point:
   1. Household and domestic wastes
   2. Industrial wastes
      (a) organic
      (b) inorganic
      (c) heat
      (d) radioactive substances
   3. Surface wastes from impermeable areas
      (a) inadvertently discharged pollutants
      (b) avoidably discharged wastes (e.g. surface waters from airports and industrial sites)
   4. Wastes from specifically drained sites
      (a) suspended solids from construction sites, quarries and mines
      (b) run off from spoil heaps
      (c) leachates from lined landfill sites
      (d) mine drainage water (dissolved pollutants)
   5. Illegal discharges (theoretically controlled by legislation but requiring monitoring and detection).

B. Diffuse sources
   1. Contaminated groundwater and natural drainage flows
   2. Dispersed matter caused by navigational disturbance (e.g. resuspended sediments)
   3. Contaminated rainfall and atmospheric fallout
   4. Accidental discharges, unpredictable in time and space.

The extent to which any one of these sources is dominant in causing pollution depends upon the nature of the catchment. Rural, sparsely populated and poorly cultivated catchments are likely to possess clean rivers and any pollution that is present will probably be caused by diffuse sources. Urban, heavily populated catchments are likely to possess polluted rivers and point sources will predominate. Depending on the integrity of the artificial drainage system, overflows from sewers and discharges from storm water culverts may be as damaging as the continuous discharge from treatment plants. The important effects attributable to the sources are summarized in Table 2.3.

2.2.1 Community waste-waters

Community waste-waters comprise discharges from households and effluents from industrial and
Table 2.3 The principal effects of pollutants characterized by source

<table>
<thead>
<tr>
<th>SOURCE</th>
<th>Potentially toxic</th>
<th>Disease transmission</th>
<th>Loss of oxygen resources</th>
<th>Eutrophication</th>
<th>Damage to food chains</th>
<th>Loss of aesthetic values</th>
</tr>
</thead>
<tbody>
<tr>
<td>POINT</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1. Household and human wastes</td>
<td>+</td>
<td>++</td>
<td>++</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>2. Industrial wastes</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(a) organic</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(b) inorganic</td>
<td>++</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(c) heat</td>
<td></td>
<td>++</td>
<td></td>
<td></td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>(d) radioactive substances</td>
<td>+</td>
<td></td>
<td></td>
<td></td>
<td>++</td>
<td></td>
</tr>
<tr>
<td>3. Surface wastes</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(a) inadvertently discharged</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(b) avoidably discharged</td>
<td>+</td>
<td></td>
<td></td>
<td></td>
<td>++</td>
<td>+</td>
</tr>
<tr>
<td>4. Specifically drained</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(a) construction site etc. drainage</td>
<td>+</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(b) spoil heap drainage</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(c) leachates</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>(d) minewaters</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>5. Illegal discharges</td>
<td>+</td>
<td></td>
<td></td>
<td></td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>DIFFUSE</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1. Contaminated groundwater</td>
<td></td>
<td></td>
<td></td>
<td>++</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>and natural drainage</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2. Navigational disturbance</td>
<td>+</td>
<td></td>
<td></td>
<td></td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>3. Atmospheric fall-out</td>
<td>+</td>
<td></td>
<td></td>
<td></td>
<td>++</td>
<td></td>
</tr>
<tr>
<td>4. Accidental discharge</td>
<td>++</td>
<td></td>
<td></td>
<td></td>
<td>+</td>
<td>++</td>
</tr>
</tbody>
</table>

++ - more important effect

commercial premises connected to the public sewerage system. Their composition is determined by the presence of human excreta, unconsumed residues from food, household cleaning agents and other products disposed of via the kitchen sink and lavatory basin. Sewage is invariably rich in bacteria and bacterial substrates and its most important property is its putrescibility which must largely be eliminated by aerobic treatment before the sewage can safely be released to surface waters. The polluting strength of sewage is usually characterized by its 'biochemical oxygen demand' (BOD). This determines whether the sewage is 'strong', 'medium' or 'weak' (see Table 2.4). Other characteristics routinely determined are suspended solids and ammoniacal nitrogen content. Where relatively large volumes of industrial waste-waters are admitted to the public sewers for treatment in admixture with domestic sewage, additional determinations of heavy metals, cyanide, oil and other substances whose presence is suspected may be made. Particular attention may be paid to heavy metal ions because of the propensity of these to become adsorbed on particulate matter and accumulated in sludge.

When expressed in terms of commonly determined characteristics the composition of sewage is found to vary from place to place and over time at any given location. The major causes of variability are: the proportion of industrial wastes and their composition; the daily volume of water customarily used per household; the dietary habits of the population and the integrity of the sewerage system in preventing excessive infiltration of groundwater.

Some average values of the composition of community waste-water in six towns in Brazil (Table 2.5) may be compared with the data given in Table 2.4 for North American sewage. The
Table 2.4 Typical composition of community waste-water, after Metcalf and Eddy, 1974. (All values except settleable solids are expressed in mg/litre)

<table>
<thead>
<tr>
<th>Constituent</th>
<th>Concentration</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Strong</td>
</tr>
<tr>
<td>Biochemical oxygen demand, 5-day, 20°C (BOD(_{5})-20°C)</td>
<td>300</td>
</tr>
<tr>
<td>Chemical oxygen demand (COD)</td>
<td>1,000</td>
</tr>
<tr>
<td>Solids, total</td>
<td>1,200</td>
</tr>
<tr>
<td>Dissolved, total</td>
<td>850</td>
</tr>
<tr>
<td>Fixed</td>
<td>525</td>
</tr>
<tr>
<td>Volatile</td>
<td>325</td>
</tr>
<tr>
<td>Suspended, total</td>
<td>350</td>
</tr>
<tr>
<td>Fixed</td>
<td>75</td>
</tr>
<tr>
<td>Volatile</td>
<td>275</td>
</tr>
<tr>
<td>Settleable solids (ml/litre)</td>
<td>20</td>
</tr>
<tr>
<td>Total organic carbon (TOC)</td>
<td>300</td>
</tr>
<tr>
<td>Nitrogen, (total as N)</td>
<td>85</td>
</tr>
<tr>
<td>Organic</td>
<td>35</td>
</tr>
<tr>
<td>Free ammonia</td>
<td>50</td>
</tr>
<tr>
<td>Nitrites</td>
<td>0</td>
</tr>
<tr>
<td>Nitrates</td>
<td>0</td>
</tr>
<tr>
<td>Phosphorus (total as P)</td>
<td>20</td>
</tr>
<tr>
<td>Organic</td>
<td>5</td>
</tr>
<tr>
<td>Inorganic</td>
<td>15</td>
</tr>
<tr>
<td>Chlorides</td>
<td>100</td>
</tr>
<tr>
<td>Alkalinity (as CaCO(_3))</td>
<td>200</td>
</tr>
<tr>
<td>Grease</td>
<td>150</td>
</tr>
</tbody>
</table>

Table 2.5 Strength of community waste-water in six communities in Brazil (after Okun and Ponghis, 1975)

<table>
<thead>
<tr>
<th>Constituent</th>
<th>Average value mg/litre</th>
</tr>
</thead>
<tbody>
<tr>
<td>BOD(_{5})-20°C</td>
<td>300</td>
</tr>
<tr>
<td>Solids, total</td>
<td>700</td>
</tr>
<tr>
<td>Solids, suspended, total</td>
<td>280</td>
</tr>
<tr>
<td>Settleable solids (ml/litre)</td>
<td>4</td>
</tr>
<tr>
<td>Grease</td>
<td>60</td>
</tr>
</tbody>
</table>

differences are not great, the sewage in Brazil corresponding to a medium strength North American sewage. A more useful basic statistic which enables the composition of sewage to be estimated on the basis of water consumption is the load of BOD and solids contributed by the population expressed as grams per person per day. A range of values is presented in Table 2.6 and typical concentrations of solids are shown in Table 2.7. It should be pointed out that these data have been derived from sewage in industrialized countries and may not be typical of developing nations. Where such data are lacking, they should be determined on a local basis to support efficient, cost-effective design of treatment facilities.

2.2.2 Industrial wastewaters

While it is possible to give some common characteristics of sewage (e.g. solids, BOD), the composition of industrial waste-waters will depend upon the process producing the aqueous wastes and the regulations governing the discharge of such wastes to public sewers. An attempt is made here to document some of the major pollutants likely to be found in industrial wastewaters.
Table 2.6 Wastes contributed to community waste-waters (after Okun and Ponghis, 1975)

<table>
<thead>
<tr>
<th>Solids g/person/day</th>
<th>BOD$_{5-20^\circ}$ g/person/day</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Inorganic</td>
</tr>
<tr>
<td>Suspended (Settleable)</td>
<td>15</td>
</tr>
<tr>
<td>Non-settleable</td>
<td>10</td>
</tr>
<tr>
<td>Dissolved</td>
<td>75</td>
</tr>
<tr>
<td>Total</td>
<td>90</td>
</tr>
</tbody>
</table>

A = Austria (Okun and Ponghis, 1975)
USA = United States of America (Fair et al. 1966)

Table 2.7 Average composition of community waste-waters

<table>
<thead>
<tr>
<th>Solids mg/litre</th>
<th>BOD$_{5-20^\circ}$ mg/litre</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Inorganic</td>
</tr>
<tr>
<td>Suspended (Settleable)</td>
<td>75</td>
</tr>
<tr>
<td>Non-settleable</td>
<td>50</td>
</tr>
<tr>
<td>Dissolved</td>
<td>375</td>
</tr>
<tr>
<td>Total</td>
<td>450</td>
</tr>
</tbody>
</table>
2.2.2.4 Discharges from pharmaceutical industries

Effluents may contain substances with a high BOD such as carbohydrates and protein in waste cultural media; organic solvents and extractants, intermediate compounds and final reaction products such as antibiotics and drugs. Pharmaceutical products are designed to have a specific chemical or biological activity and this does not cease when they pass from person, plant or animal intended for primary use. Although discharged in low concentrations, the property of bioaccumulation over a long period of time should not be forgotten.

2.2.2.5 Discharges from soap and detergent industries

Effluents may contain fatty acids and alcohols, fats, oils, glycerine, sulphonated hydrocarbons and polyphosphates. Most of these are biodegradable and some, particularly the polyphosphates are important plant nutrients which give rise to eutrophication when discharged in excess. Other compounds such as alkyl-aryl sulphate and tertiary ammonium compounds are non-degradable or toxic.

2.2.2.6 Discharges from food processing industries

Effluents are highly variable with a BOD ranging from $10^2$-10$^5$ mg/litre (Farrimond 1980). The pH can be acidic (milk wastes) or alkaline (caustic washing of potatoes). High BOD values are found in the effluents of fruit and vegetable processes. Sugar beet wastes contain large amounts of dissolved sucrose and suspended solids. Wastes from dairies, abattoirs and breweries are all characterised by a high BOD caused by carbohydrates and proteins. Effluents from such industries when discharged to rivers can cause excessive growths of sewage fungus. Most effluents are treatable in admixture with domestic sewage.

2.2.2.7 Discharges from paper manufacturing industries

Effluents may contain large quantities of cellulose fibres and wood sugars with a high BOD, lignosulphonic acid (which is non-degradable), organic acids, sulphites and sulphates. Effluents are often discharged into a water course and this may give rise to growths of sewage fungus and a reduction in the aesthetic value of the receiving water.

2.2.3 Pollutants from agriculture

Since the second world war there has been an intensification of the production of many crops and farm animals. This has been brought about by: increased use of agrochemicals (fertilizers and pesticides); increased mechanization and increased specialization. Most pollutants arise from the increased use of agrochemicals.

2.2.3.1 Fertilizers

The application of artificial fertilizer has increased substantially in most developed countries over the last 50 years. The effect of increased nitrogen usage is reflected in increasing concentrations of nitrates in rivers, lakes and groundwaters. The annual mean concentration of NO$_3$-N in the River Thames (Walton) in 1950 was around 5 mg/l but in 1976 the mean concentration was around 15 mg/l. A doubling in the nitrate concentration from 5 to 10 mg/l has been observed in borehole water from the Triassic sandstone of Worcestershire (UK). Part of the explanation for these increases is the extra quantities of nitrogenous fertilizers used by farmers but the change from grassland to arable land is also implicated. There is adequate evidence to show that the drainage water from under arable land contains a great deal more nitrate than that from grassland. The relevance of this to pollution is the recommendation of WHO not to supply water in excess of 11.3 mg NO$_3$-N/litre because of the possibility of causing methaemoglobinemia in infants. A further factor is the role of inorganic nitrates in stimulating excessive plant growths in lakes and reservoirs (eutrophication). However, this can only occur if inorganic phosphates are also present.

In most undeveloped upland catchments, the concentration of inorganic phosphate in waters draining them is very low, usually a few micrograms per litre. Inorganic phosphate is retained by most agricultural soils and evidence points to very little fertilizer derived phosphorus moving into drainage waters or deep aquifers in the way that nitrate is. The amount of phosphorus in drainage water is controlled by soil type rather than rates of application. It is only on soils containing little or no clay component, e.g. light sandy or peat soils, that substantial losses of phosphorus occur. It is significant to note that increased afforestation in upland areas with mainly peat soils may well bring about significant increases in phosphorus concentration in waters draining from such areas. This is particularly important in the United Kingdom because the upland regions are also the main gathering grounds for water and eutrophication problems in upland reservoirs are likely to occur. Concentrations as low as 0.01 mgP/litre have been shown to stimulate excessive growths of algae in lakes and reservoirs (Youngman and Lack 1981).

2.2.3.2 Pesticides

The use of chemicals to control weeds, pests and diseases has increased enormously from 1950 onwards. The term pesticide embraces: insecticides, acaricides, nematicides, molluscicides,
herbicides, fungicides, soil fumigants and growth regulators. So far as water quality is concerned consideration has to be given to the application and movement of pesticides, their degradation products or their persistence in the environment and the importance of these to ecology, including man.

Probably the most persistent pesticides are the organochlorine compounds whose traces are to be found throughout the world. During their passage through the food chains these chemicals are concentrated in the fatty tissues and for this reason, their use is banned in many countries. The rate of introduction of new pesticides is slowing down, largely because of the great cost and increased commercial risk. Manufacturers now look for pesticides which will control a broad spectrum of organisms rather than highly selective ones for particular pests. Such developments will inevitably lead to greater problems in freshwaters when accidental contamination occurs. From time to time dramatic episodes are caused by the unintentional entry of pesticides into watercourses. Often these are caused by ignorance, carelessness or mechanical failure and these incidents should not be regarded as indications of the real dangers from the use of such chemicals.

2.2.3.3 Intensive livestock farming

Increasing the intensity of livestock farming increases the problem of animal slurries as potential pollutants. Lowland beef and dairy cattle are normally housed for part of the year and usually the slurry that accumulates during that period can be disposed of to the land used for grazing. Pigs and poultry may be housed in intensive units and may pose serious pollution problems because the conditions can demand more land than is available for slurry disposal. Animal slurry is characterized by its high BOD, high concentrations of nitrogenous compounds, phosphates and suspended solids. Even when successfully applied to land the slurry may create problems because disposal is often necessary during the wet season when soil leaching and wash-off is most intensive.

Associated with the more intensive livestock farming is the increase in the use of silage as a winter feed and silage liquors with a BOD in excess of $10^6$ mg/litre occasionally cause local pollution problems.

2.2.4 Other potential sources of pollution

There are no general characteristics to be attached to these sources and a case by case approach is probably the best way of dealing with their analysis. Generally, the polluting substances do not differ from the ones previously discussed.

2.2.4.1 Atmospheric pollution

There is particular concern internationally with the problem of acid rain. This is produced when atmospheric moisture combines with the oxides of sulphur and nitrogen emitted when fossil fuels are burnt at power stations, factories and vehicles. The effects of acid rain are neutralized in chalk and limestone districts and are severe in regions where the natural rocks have no buffering capacity. The ecological effects are manifested by a marked decline in the variety of invertebrate animals and absence of fish species at about pH5. Waters below pH4 are usually lethal to salmonid fish. As the acidity of lakes increases there is usually a marked change in the flora.

Sodium, potassium, magnesium and chloride in rain are derived principally from the oceans and show a decreasing gradient from west to east over Europe.

Nitrate in rain is land derived and shows considerable geographical differences. For example the deposition of nitrate nitrogen from the atmosphere varies from 20 kg/ha in eastern Britain to 60 kg/ha in the Shetland Isles north of Scotland.

Minor elements in aerial deposition include the toxic heavy metals although the quantities are not significant by comparison with other sources.

2.2.4.2 Trunk highway drainage

Drainage from trunk highways (motorways, autobahns, autostradas, etc.) is a possible source of minor local problems but the current design of drainage systems appears to provide adequate control. Pollution is largely attributable to suspended solids, oil/rubber/water emulsions and to road salt applied in winter.

2.2.4.3 Leachates from landfill waste disposal sites

Special hazardous wastes such as asbestos, halogenated and other solvents, mineral oil, pharmaceuticals, tarry wastes, tanning wastes and metal finishing waste are often co-disposed with household waste at licensed landfill sites. The decomposition process is anaerobic and slow and leachates may include ammonia, volatile fatty acids, chlorides, sulphites and sulphates.

REFERENCES


3 Characteristics of river systems
3.1 INTRODUCTION

In addition to being major sources of water supply, rivers are used as the principal disposal pathways for waste materials. As society becomes more industrialized the variety of these waste materials increases and in many countries the problems of water quality are more difficult and demanding than water quantity. In order to manage a river system and maintain adequate water quality it is essential, therefore, to understand the mechanisms governing pollution and self purification processes. The interactions between physical, chemical and biological processes ensure that forecasting the impact of an effluent on a river system is particularly difficult. Figure 3.1 indicates the effects of an organic effluent on a river and the changes occurring downstream from the outfall. Self-purification processes depend on a wide range of parameters and, in this chapter, the physical, chemical and biological mechanisms governing water quality in river systems are considered.

3.2 PHYSICAL AND HYDRAULIC BEHAVIOUR

3.2.1 Water velocity and streamflow regimes

Any substance present in flowing water moves with the current. The transport involved is called the advective transport of the substance and its direction coincides with that of the current. This movement of water in rivers is accomplished primarily under the effect of gravity. However additional effects can also be exerted in large rivers by, for example, the forces in the curves of a river section and by such factors as the wind.

The velocity of water movement increases with the slope of the river and, hence, the velocity in the upper reaches of a river is generally greater than that in the lower reaches where the slopes are not so steep. This fact usually creates conditions for the scouring of solid material from the river bed in mountainous regions and the transport and deposition of this material downstream. These three processes (erosion, transport and accumulation) can be observed at one and the same time in river reaches and a river bed is considered to be relatively stable when erosion and sedimentation are in balance. Detailed descriptions of sedimentation processes are given elsewhere (Unesco, in press) and it is sufficient here to note that on account of erosion sinuous sections of the river bed are formed together with river meanders. In the sinuous sections of the river bed a transverse circulation is created. From the combination of forces the water particles in the sinuous sections move in a spiral fashion, first turning towards the concave bank, then sinking towards the bottom provoking erosion of the concave side. The circulation patterns increase water turbulence and enhance the dispersion of pollutants, as indicated in Figure 3.2.

The distribution of water velocity in a river cross-section shows great diversity and, in general, the following conditions apply:

1. velocity is smallest near the bottom and the banks, where the resistance of the river bed roughness is the greatest;
2. velocity increases from the bottom and the banks towards the middle of the river and towards the surface, being at its maximum when it is slightly under the water surface or on the surface itself;
3. when the surface is occupied by an ice cover a resistance occurs from friction on the ice cover which causes the maximum velocity on the vertical to be lower than if it were on a free water surface.
Figure 3.1 Changes in stream quality downstream of a waste outfall (Unesco/WHO, 1978, Water Quality Surveys)
Figure 3.2 Water movement in a stream (a) cross-section, and (b) in a curve
Despite the apparent complexity of water velocities in rivers it is still possible to define an overall mean velocity for movement of pollutants in river systems. For example, Figure 3.3 shows the results of an experiment on a river in which a mass of tracer is discharged instantaneously and the tracer concentration is monitored at a downstream point on the river. The pure transportation delay followed by the 'bell shaped' concentration curve is typical for river systems. The mean travel time down the river may be determined by calculating the time corresponding to the arrival of the centre of gravity of the curve at the monitoring point. The mean velocity is determined from the mean travel time, $T$, and the distance between injection and monitoring locations, \( u = \frac{L}{T} \) (3.1)

It is interesting to note that hydrologists occasionally use the tracer technique to determine stream flow, $Q$, since

\[ Q = \frac{M}{\int_0^\infty c \, dt} \]  

where $M$ is the mass of tracer injected, and $\int_0^\infty c \, dt$ is the integrated area under the concentration curve. Along curved sections of river, however, dispersion patterns are often complex (Figure 3.4).

An accurate estimate of streamflow is essential for a quantitative analysis of pollution problems since the streamflow determines the dilution properties of a river. However streamflow is not a constant quantity and varies significantly throughout the year. Climatic and geographic variations give rise to changes in hydrograph shape (IAHS/Unesco/WHO, 1967) and periods of high flow and low flow exert a major influence on the processes of self-purification of river water. In periods of high flow, for example, the natural aeration processes are enhanced by the increased turbulence and self-purification is improved. During low flow periods dilution of pollutants is reduced, self-purification processes are retarded and sediments settle on the river bottom together with adsorbed pollutant matter. Intensive rain following periods of low flow also creates problems by increasing stream velocity and scouring of oxygen-demanding organic matter from the river bed. A detailed description of oxygen balance processes is given later in this chapter.

In large rivers with comparatively shallow water and low water velocities the natural mixing processes are reduced and non-homogeneity of the chemical composition of water is observed. Parallel currents of different chemical composition may form downstream of a tributary joining the river or the inflow of waste into the river. This is illustrated below the confluence of the River Mosel and the River Rhine, as shown in Figure 3.5, where the darker Mosel water is clearly separated from the Rhine water for some distance below the confluence. Such heterogeneity in the natural chemical composition has been observed in many large rivers, e.g. the Mississippi after the Missouri's inflow, the River Elbe after the entry of the Saale and the River Don after Donetz. The problems of estimating the mixing characteristics of a river are considered later in this chapter.

3.2.2 Dispersion of pollutants

Pollutants discharged into rivers whether from point or non-point sources are subject to the processes of dispersion. In the tracer experiment, illustrated in Figure 3.3, the 'bell shaped' nature of the curve is an indication of the dispersive properties of the stream. The dispersion process represents an important factor when assessing pollution problems.

3.2.2.1 Molecular diffusion

The spreading of a pollutant due to the motion of the molecules at the interface with the receiving fluid is termed molecular diffusion. The molecular motion at the interface is random in character and the molecules of the pollutants are constantly colliding with those of the receiving body of water. Because of these collisions, the molecules of the pollutant move sometimes towards a region of higher concentration and sometimes towards that of a lower concentration without having preference for one or the other. This type of motion of a molecule can be identified as one of the stochastic processes, namely the 'random walk'. Even though there is no preferred direction of motion of molecules performing a random walk, a net transfer of the molecules of the pollutants from the region of higher concentration to that of lower concentration can be noticed.

The spreading of substances due to the random motions of the molecules was expressed for the first time in quantitative terms by Fick (1855), who made an analogy between this process and the process of transfer of heat by conduction which is also due to the random motion of the molecules and adopted the mathematical equation of heat conduction derived earlier by Fourier (1822), to express the amount of substance diffusing through unit area of a section. According
Figure 3.3 Tracer concentration curve in the Bedford Ouse River. Experiment conducted along an 8 kilometre reach of river and tracer injected at 12.30 on 21 January 1980 (Whitehead, 1981)
Figure 3.4 Water movement in a curve of a river (a) depth, (b) velocity and (c) tracer concentration (*calculated concentrations)
Figure 3.5 Confluence of the River Mosel and River Rhine
to Fick, the rate of transfer of diffusing substance through a unit area of section is proportional to the concentration gradient measured normal to the section, i.e.

\[ F_L = -D \frac{dc}{dx} \]  (3.3)

where \( F_L \) is the amount of substance diffusing per unit area of a section per unit time, \( C \) is the concentration of the diffusing substance, \( x \) is the distance measured normal to the section and \( D \) is the diffusion coefficient. The minus (-) sign is due to the fact that the transfer of substance takes place in a direction along the decreasing concentration of the diffusing substance.

The molecular diffusion process plays a significant role in laminar flows to spread pollutants; but in the case of turbulent flows its effects are negligible in comparison to the 'turbulent diffusion process' which will be described in the next sub-section. Since, in almost all the natural streams the flows is turbulent, the molecular diffusion process is not a dominant one in considering the spreading of pollutants in natural streams.

3.2.2.2 Turbulent diffusion

The spreading of pollutants due to the random motion of the fluid 'particles' in a turbulent flow is termed turbulent diffusion. Turbulent diffusion is, in fact, a characteristic feature of turbulent flows. Indeed, in the classic experiment of Reynolds this characteristic behaviour indicated the onset of turbulence in a flow field. In the case of molecular diffusion process, the molecules of the diffusing substances are performing the random motions, whereas in the case of turbulent diffusion process, the 'fluid particles' are performing random motions. By 'fluid particle', we mean the volume of fluid which is so small that, within the framework of the theory of continuum, it can be identified with a point which is displaced together with the surrounding fluid. In order to trace the motion of the fluid particles, we assume the presence of some admixtures in the flow field which behave in exactly the same way as the fluid particles but have special properties to permit the tracing of their motions.

The analogy between the motion of the molecules and that of the fluid particles enables a quantitative formulation of the turbulent diffusion process. Indeed, the amount of substance \( F_t \) diffusing per unit area of a section per unit time due to turbulent diffusion is expressed in terms of a (turbulent) diffusion coefficient \( \varepsilon \) and the concentration gradient of the diffusing substance as:

\[ F_t = -\varepsilon \frac{dc}{dx} \]  (3.4)

Spreading of a substance due to turbulent diffusion is much more rapid than that due to molecular diffusion. In other words, the numerical values of \( \varepsilon \) are far greater than those of \( D \).

3.2.2.3 Secondary circulation

In addition to spreading due to the random motion of molecules and fluid particles, the pollutants are also carried by the time averaged convective velocities. Therefore, an accurate knowledge of the time average velocity field is a prerequisite to determining the fate of the pollutant introduced into the flow field. In non-circular flow cross-sections, typical of natural streams, it has been observed that there is a mean current in a plane perpendicular to the direction of the flow even for a straight channel. Such currents, called the secondary current of the second kind by Prandtl (1952), can carry the pollutants in the transverse plane of the flow cross-section.

At present, the knowledge of this secondary current is limited. The usual practice in dispersion studies is to lump the effects of secondary currents with the other diffusion processes and measure an overall dispersion coefficient. Recent investigations of the spreading of pollutants in the transverse direction in channels with secondary circulation have revealed that the effects of secondary circulations on the spreading of pollutants are far greater than those of the turbulent and molecular diffusion processes and hence have demonstrated a need for further research on this type of secondary circulation.

3.2.2.4 Settling and transport with sediments

If the pollutant contains particulate matter or if it coats the sediment particles present in the alluvial streams as a result of the physico-chemical processes such as adsorption, then the transport and spreading of the pollutant are governed by the mechanisms controlling the motion of the sediment particles (Unesco, in press).

The motion of sediment particles has been studied extensively in the field of sediment transport and a brief account is given here. The strength of the flow needed to initiate the motion of the sediment particles forming the bed of the alluvial stream depends on the size and
the specific weight of the particles and can be predicted with a reasonable accuracy using the well-known Shields Curve shown in Figure 3.6. In this figure, the critical mobility number $Y_{cr}$ is plotted against the critical grain size Reynolds Number $X_{cr}$. The mobility number $Y$ expresses the balance between the shear stress ($T$) exerted by the flow on the bed and the weight of a solid particle per unit surface area and hence can be written as:

$$Y = \frac{\rho \gamma_{*}^2}{\gamma_s P}$$

(3.5)

where $\rho$ is the density of the fluid, $\gamma_*$ ($=\sqrt{\frac{T}{\rho}}$) is the shear velocity of the flow, $\gamma_s$ is the submerged specific weight of the solid particle and $P$ is the size of the particles.

The grain size Reynolds Number $X$ expresses the balance between the inertia and the friction forces and can be given by:

$$X = \frac{\rho \gamma_* P}{\mu}$$

(3.6)

where $\mu$ is the dynamic viscosity of the fluid.

The values of $Y_{cr}$ and $X_{cr}$ are those resulting from the use of critical values of $\gamma_*$ (i.e. $\gamma_{*_{cr}}$) in equations (3.5) and (3.6). When the mobility number $Y$ for a flow is less than the critical mobility number $Y_{cr}$ then the particles will not be moved by the flow and under these circumstances the bed of the stream can be treated as a rigid one. If the strength of the flow increases, increasing the shear stress at the bottom and consequently the mobility number $Y$, the particles begin to move. If the ratio $Y/Y_{cr}$ is between 1 and 15, the particles move in the vicinity of the bed by rolling and performing short jumps. During this stage of the flow, sand dunes begin to develop at the bottom and the particles advance along the bed in a series of alternate rest and transport periods. Experiments indicate that the rest and the transport periods are random in nature and hence the sediment particles are dispersed as they move with the flow. Shen and Cheong (1973) have formulated the dispersion of the contaminated "Bedload" particles using the experimental evidence that the rest periods are exponentially distributed and the step lengths (the length of travel of particles between rest periods) are gamma distributed.

For values of $Y/Y_{cr}$ greater than 15, the dunes begin to disappear and the particles are entrained into the main body of the flow, moving in suspension in irregular and random paths like the fluid particles. Dispersion of the sediment particles during this stage of motion can be formulated in the same way as for the fluid particles, taking into account the negative buoyancy of the sediment particles.

So far in this section various hydrodynamical processes which are responsible for the spreading of the pollutants have been considered in a descriptive manner. In the following, a quantitative treatment of the transport of pollutants in a natural stream, where all these processes can interact, will be considered.

### 3.2.3 Mass-balance equations

#### 3.2.3.1 General mass-balance equation

Mathematical treatment of the transport of pollutants in streams is based on the principle of conservation of mass. In the following derivation of the mass-balance equation for a pollutant, it is considered for simplicity that the pollutant is a conservative substance. By the term 'conservative substance' we mean those substances which do not undergo any process other than dilution. In other words, the processes such as decay, chemical reactions and production or removal are not considered. Consideration of the non-conservative substances will be taken up later on in this chapter.

Consider a closed volume $V$ as shown in Figure 3.7 and let $F$ be the rate of transfer of a diffusing substance per unit area through the surface $S$ enclosing the volume $V$. The conservation of the mass of the diffusing substance entering and leaving the volume $V$ per unit time must be equal to the rate of change of the concentration $C$ of the diffusing substance in $V$ at time $t$.

This mass conservation principle leads to the commonly used mass balance equation which for a neutrally buoyant and conservative substance, can be expressed as:

$$\frac{\partial C}{\partial t} + u_x \frac{\partial C}{\partial x} + u_y \frac{\partial C}{\partial y} + u_z \frac{\partial C}{\partial z} = \frac{\partial}{\partial x} \left[ D \frac{\partial C}{\partial x} \right] + \frac{\partial}{\partial y} \left[ \epsilon_d \frac{\partial C}{\partial y} \right] + \frac{\partial}{\partial z} \left[ \epsilon_d \frac{\partial C}{\partial z} \right]$$

(3.7)

It is assumed that the molecular diffusion coefficient $D$ is constant in all directions whereas the turbulent diffusion coefficient, $\epsilon_d$, depends on the co-ordinate direction and can be expressed by three scalars, $\epsilon_{dx}$, $\epsilon_{dy}$ and $\epsilon_{dz}$.

For the case of sediment particles which are heavier than water and which have a fall
Figure 3.6 Shields Curve for sediment transport
Figure 3.7 Control volume in a flow field
velocity of \( w \) in the \( y \) direction, the mass balance equation can be written as:

\[
\frac{\partial C}{\partial t} + u_x \frac{\partial C}{\partial x} + (u_y - w) \frac{\partial C}{\partial y} + u_z \frac{\partial C}{\partial z} = \frac{1}{\partial x} \left[ (D + \epsilon_x) \frac{\partial C}{\partial x} \right] + \frac{1}{\partial y} \left[ (D + \epsilon_y) \frac{\partial C}{\partial y} \right] + \frac{1}{\partial z} \left[ (D + \epsilon_z) \frac{\partial C}{\partial z} \right]
\] (3.8)

For reasons indicated earlier, if the effect of the molecular diffusion is negligible in comparison to that of turbulent diffusion, then the molecular diffusion coefficient \( D \) can be left out of both equations (3.7) and (3.8).

In order to solve the mass balance equation (i.e. equation (3.7) for neutrally buoyant pollutants and equation (3.8) for negatively buoyant pollutants) to predict the concentration of the pollutant as a function of space \((x, y, z)\) and time \( t \), the velocity components \( u_x, u_y \) and \( u_z \) and the diffusion coefficients at every point in the flow field have to be known with the appropriate initial boundary conditions for concentration distribution. Such a detailed description of the velocity field and the diffusion coefficients is usually not available for the natural streams and therefore, the mass balance equation is often simplified by making certain assumptions regarding the flow field and the diffusion processes. Some of the commonly made simplifications will be considered here.

3.2.3.2 Longitudinal dispersion: cross-sectional average mass-balance equation

Taylor (1954) described the spreading of a neutrally buoyant, conservative pollutant in a unidirectional shear flow (i.e. flow in pipes) by making the following assumptions:
1. The motion is predominantly in one direction, i.e. \( u_y \) and \( u_z \) are zero;
2. The turbulent diffusion in the \( y \) and \( z \) directions of motion is negligible, i.e. \( \epsilon_y = 0 \), \( \epsilon_z = 0 \);
3. The molecular diffusion is negligible, i.e. \( D = 0 \).

By averaging over the entire cross-section and writing

\[
u_x = \bar{u} + u'
\]

\[
\bar{C} = \bar{C} + C'
\]

where the overbar denotes averages over the cross-section and \( u' \) and \( C' \) represent spatial variations from the mean, equation (3.7) can be written as:

\[
\frac{\partial \bar{C}}{\partial t} + \bar{u} \frac{\partial \bar{C}}{\partial x} = E \frac{\partial^2 \bar{C}}{\partial x^2}
\] (3.10)

which is the well-known Taylor's one-dimensional dispersion equation. \( E \) is an apparent longitudinal diffusion coefficient which will henceforth be referred to as the longitudinal dispersion coefficient. Theoretical approaches to the estimation of the dispersion coefficient are given by Elder (1959) and Fisher (1966, 1967).

In a uniform channel with constant stream velocity, the dispersion coefficient may be calculated from a tracer curve as

\[
E = \frac{a^2u^3}{2L}
\] (3.11)

where \( L \) is the distance travelled, \( \bar{u} \) is the velocity and \( a^2 \) is the variance of the concentration curve of the type shown in Figure 3.3.

It should be pointed out that Taylor's one-dimensional approach does not apply in regions close to the location where the pollutants are injected. Fisher (1966) demonstrates that the one-dimensional approach of Taylor is valid only when \( C' \) is much smaller than \( \bar{C} \) and identifies two distinct periods in the transport of the pollutants from a point source in a natural stream. These two periods are:
1. The convective period, during which the movement of the pollutants is dependent on the initial convective velocity and the concentration distribution is highly skewed;
2. The diffusive period, during which the bulk motion of the pollutants can be described by Taylor's one-dimensional equation.

Based on his experimental results, Fisher found that the time \( T \) at which the diffusive period began was about 0.4 times the Eulerian time scale for the flow, i.e.

\[
T = 0.4 \frac{a^2}{\epsilon_z}
\] (3.12)

where \( a \) is the distance between the point of maximum velocity to the most distant bank.

In terms of distance \( L_c \) downstream from the source of pollution, the criterion for the use of Taylor's equation becomes:

\[
L_c > \frac{1.8a^2 \bar{u}}{Rv_u}
\] (3.13)
3.2.3.3 Transverse dispersion: depth-average mass-balance equation

Since in most natural streams, width is considerably larger than the depth, the pollutants become uniformly spread over the depth much faster than they are spread uniformly across the width and hence it is customary to treat the motion of the pollutants in terms of the depth average version of the mass balance equation. E. R. Holley (1971) integrated equation (3.7) with zero molecular diffusion over the depth and obtained the following equation:

\[
\frac{\partial C}{\partial t} + u_x \frac{\partial C}{\partial x} + u_z \frac{\partial C}{\partial z} = \frac{\partial}{\partial x} \left( \frac{\partial C}{\partial x} \right) + \frac{\partial}{\partial z} \left( \frac{\partial C}{\partial z} \right)
\]  

(3.14)

where the overbar now denotes the average over the depth and the dispersion coefficients \( e_x \) and \( e_z \) are defined by the following expressions:

\[
\overline{u_x'C'} - \overline{\varepsilon_x \frac{\partial C}{\partial x}} = e_x \frac{\partial C}{\partial x}
\]

(3.15)

\[
\overline{u_z'C'} - \overline{\varepsilon_z \frac{\partial C}{\partial z}} = e_z \frac{\partial C}{\partial z}
\]

\( e_x \) and \( e_z \), therefore, include the effects of both the turbulent diffusion and those of the differential convection.

For a continuous injection of the pollutant when the concentration distribution becomes independent of time \( t \), the longitudinal dispersion of pollutants is often neglected because of negligible concentration gradient in the longitudinal direction and hence the equation representing such a case becomes:

\[
\frac{\partial C}{\partial t} + u_x \frac{\partial C}{\partial x} = \frac{\partial}{\partial x} \left( \frac{\partial C}{\partial x} \right) \left( h \varepsilon_x \right) \frac{\partial C}{\partial x} \]  

(3.16)

If the channel reach is straight, \( \bar{u}_z \) becomes zero and if \( h \) is constant equation (3.16) takes the form:

\[
\overline{u_x'C'} = \overline{\varepsilon_x \frac{\partial C}{\partial x}} = e_x \frac{\partial C}{\partial x}
\]

(3.17)

\( e_x \) for this case, includes the effects of turbulent diffusion in the \( z \) direction and the effects of the secondary currents of the second kind which result from the non-circular geometry. If the channel reach is a meandering one, then \( u_z \) becomes the secondary current resulting from the curvature (secondary currents of the first kind in Prandtl's terminology) and does not vanish. \( u_z \) can be computed from the depth average continuity equation for incompressible flows at:

\[
u_z = -\frac{1}{h} \frac{\partial}{\partial x} \int_0^h \bar{u}_z \, dh
\]

(3.18)

in terms of the depth average longitudinal velocity \( \bar{u}_x \) and the flow depth \( h \). The depth \( h \) also varies along the channel either due to superelevation in the case of rigid bottom channels or due to the scour and deposition of sediments in alluvial channels. The dispersion coefficient \( e_z \) for this case includes the effects of turbulent diffusion and the differential convection resulting from the secondary current of the first kind. Therefore, the full form of equation (3.16) is necessary to describe the transport of the pollutants in meandering reaches of the stream. The dispersion coefficient \( e_z \) in both cases has to be measured experimentally since there is no theoretical means to predict it at the present time.

Several investigators have measured \( e_z \) under different channel geometry and flow conditions and a summary of all the existing measurements can be found in a paper by Krishnappan and Lau (1974).

It should be remembered that the depth average mass-balance equation (i.e. equation (3.16)) is not valid in regions where the pollutants are not mixed completely over the depth. Sayre (1973) has established from a theoretical solution of the mass-balance equation in 2-dimensional flow that the distance \( L_d \) from the point of injection where equation (3.16) is not valid is:

\[
L_d = 0.5 \left( \frac{h}{2} \right)^2 \cdot \frac{\bar{u}_x}{\varepsilon_y}
\]

(3.19)

where \( \varepsilon_y \) is an average turbulent diffusion coefficient in the vertical direction. Equation (3.19) is applicable for the case when the pollutant is injected at the mid-depth.

The vertical turbulent diffusion coefficient \( \varepsilon_y \) is obtained by assuming that the turbulent transfer of mass and momentum are equivalent, that the shear stress in the vertical is
linearly distributed and that the velocity distribution is logarithmic. The expression for \( \bar{e}_y \) under the above assumption becomes:

\[
\bar{e}_y = 0.07 \ h \cdot v_*
\]  

(3.20)

With this value of \( \bar{e}_y \), equation (3.19) reduces to:

\[
\frac{L_d}{h} = 1.8 \ \frac{u_x}{v_*}
\]  

(3.21)

Within the region between the section where the pollutant is injected and the section which is at a distance \( L_d \) downstream, one has to treat the transport of the pollutants in terms of the full mass balance equation, i.e. equation (3.7).

3.2.3.4 Non-conservative pollutants

In the case of non-conservative pollutants, the mass-balance equation (3.7) has to be corrected for the production or the removal of the pollutants due to chemical reactions, change of state or transfer across the boundaries by adding an appropriate source or sink term \( S \) which is, in general, a function of space, time and the concentration of the pollutant itself. In other words, the mass-balance equation for a non-conservative pollutant becomes:

\[
\frac{\partial c}{\partial t} + u_x \frac{\partial c}{\partial x} + u_y \frac{\partial c}{\partial y} + u_z \frac{\partial c}{\partial z} = \frac{\partial}{\partial x} (D + \varepsilon_x) \frac{\partial c}{\partial x} + \frac{\partial}{\partial y} (D + \varepsilon_y) \frac{\partial c}{\partial y} + \frac{\partial}{\partial z} (D + \varepsilon_z) \frac{\partial c}{\partial z} + S(x; y; z; t; C)
\]  

(3.22)

The term \( S(x; y; z; t; C) \) depends on the nature of the production or the removal processes but in all cases it expresses the rate at which a pollutant is added or removed from the control volume. Some of the common examples of the non-conservative substances are dissolved oxygen and biochemical oxygen demand, which will be considered in detail in the next section on the processes affecting biological self-purification.

3.3 SELF-PURIFICATION AND THE OXYGEN BALANCE

Biological self-purification is the process in which organic wastes are broken down by the respiration of micro-organisms into stable end products. It is a biochemical oxidation process through which organic wastes are consumed leaving behind end products such as carbon dioxide, water, phosphates and nitrates. The water is 'purified' in the sense that the concentration of waste material has been reduced. Organic materials which can be broken down (i.e. are biodegradable) include natural materials such as simple sugars, starch, fats, proteins as well as more complex natural or synthetic compounds which are found in sewage or other wastes. This type of respiration, in which biochemical oxidation takes place using the free dissolved oxygen in the water, is called aerobic respiration. The micro-organisms involved are called aerobes.

There is another type of respiration which can take place when free oxygen is not available. Certain micro-organisms known as anaerobes and some known as 'facultative aerobes' can still break down simple sugars without the use of oxygen. This is called anaerobic respiration. However, when anaerobic respiration takes place, end products such as hydrogen sulphide, ammonia and methane which may be toxic as well as foul smelling are often produced.

Thus, through the biochemical respiration, self-purification of the river takes place but in the same process oxygen is being removed from the aquatic environment. If the supply of oxygen into the system is exceeded by the demand from respiration, an undesirable anaerobic state will occur. Therefore, the self-purification process is very closely tied with the dissolved oxygen content and indeed with all the sources and sinks of oxygen in a river. Since dissolved oxygen is such a critical water quality parameter, the various processes affecting the dissolved oxygen will be discussed.

One of the first studies of the DO balance was by Streeter and Phelps (1925), who developed the classic oxygen balance equations by considering biochemical oxidation as the only sink and atmospheric reaeration as the only source of oxygen. These equations express the effects of BOD satisfaction or deoxygenation and reaeration through the water surface, the rates of which are expressed by rate coefficients \( K_1 \) and \( K_2 \).

The equations of Streeter and Phelps were found to be adequate to give a schematized representation of the variation of dissolved oxygen concentration downstream of the point of discharge. If the water's oxygen content is examined along the river, downstream from an assumed constant waste discharge, we notice what is known as a sag curve (see Figure 3.8). Firstly, the oxygen content decreases rapidly, reaches a minimum, then increases to reach asymptotically the saturation point. From a practical point of view, what matters is the minimum content value.

Subsequent studies have pointed out that other sources and sinks have to be taken into
Figure 3.8 The dissolved oxygen sag curve
account (Dobbins, 1964; Camp, 1963; Owens and Edwards, 1963; Edwards and Owens, 1965). A list of these processes is as follows:
- the removal of BOD by sedimentation or adsorption;
- the addition of BOD by the scour of bottom deposits or by the diffusion of partly decomposed organic products from the benthic layer into the water above;
- the addition of BOD by local runoff;
- the removal of oxygen from the water by diffusion into the benthic layer to satisfy the oxygen demand in the aerobic zone of this layer;
- the removal of oxygen from the water by the purging action of gases rising from the benthic layer;
- the addition of oxygen by the photosynthetic action of plankton and fixed plants;
- the removal of oxygen by the respiration of plankton and fixed plants;
- the continuous redistribution of both the BOD and the oxygen by the effect of longitudinal dispersion, particularly when the waste loads vary suddenly (e.g. as a result of an accident);
- the addition of oxygen by reaeration processes.

Bennett and Rathbun (1971) have written a very good summary describing the methods available to measure or evaluate the various source-sink terms. In general, the dominant processes which ought to be considered in the oxygen balance of a river are the following:
1. The BOD (biochemical oxygen demand) of the carbonaceous and nitrogenous wastes in the water;
2. Oxygen demand of the bottom deposits;
3. Immediate chemical oxygen demand;
4. Oxygen required for plant respiration;
5. Oxygen produced by plant photosynthesis;
6. Oxygen gained from atmospheric reaeration.

Referring to equation (3.22) for the mass-balance of a non-conservative substance, the equation for the dissolved oxygen (DO) distribution for the case of a straight uniform reach and negligible molecular diffusion is:
\[
\frac{\partial C}{\partial t} + u \frac{\partial C}{\partial x} = \frac{\partial}{\partial x} \left( \phi \frac{\partial C}{\partial x} \right) + \sum S_i
\]

where \(C\) is the depth average concentration and \(S_i\) represents the individual source and sink terms which will be described below.

### 3.3.1 Biochemical oxygen demand (BOD)

The BOD is the oxygen required by the micro-organisms for the respiration process described previously. The amount of the organic waste is measured by its demand on the oxygen resource. The value for the BOD concentration is not constant but changes with time and distance along a river because as oxygen is used up to oxidize the waste, the amount of waste material decreases and the oxygen demand drops. When all of the waste has been oxidized the BOD then becomes zero. BOD can also be removed from the water by sedimentation or it can be added to the water by the scouring and resuspension of bottom deposits. These factors have to be considered in order to solve for the BOD distribution along the river.

It is usually assumed that the rate of biochemical oxidation is a first order process, i.e. the oxidation rate is proportional to the amount of waste remaining in the water. The net rate of sedimentation and scour is also generally assumed to be proportional to the amount of BOD present. Therefore, referring to equation (3.22) again, the equation for the distribution of BOD can be written as:
\[
\frac{\partial L}{\partial t} + u \frac{\partial L}{\partial x} = \left( \phi \frac{\partial L}{\partial x} \right) - K_1 L - K_3 L
\]

where \(L\) = average BOD concentration in the water; \(K_1\) = rate constant for the biochemical oxidation; \(K_3\) = rate constant for the net rate of settling and resuspension of BOD.

The oxygen sink term in equation (3.23) representing the biochemical oxidation is \(-K_1 L\) and has to be obtained from the solution of equation (3.24). The amount of BOD in a sample of water has to be determined from a BOD test. The standard procedures for the test are described by the American Public Health Association (1971). In principle, this test consists of taking two samples of the same water, determining the DO concentration of one sample, incubating the other sample in darkness for five days and then determining its DO concentration. The difference in DO is the 5-day BOD of the sample. However, it is not known how well this laboratory value

\(^1\) Especially important in shallow streams with a depth of say below 2 m (Edwards and Rolley, 1965).
of BOD represents the true BOD value in the stream because the conditions of temperature, turbulence and biological activity are all different. Hull (1969) and Camp (1965) have recommended using a dark bottle suspended in the river for the BOD test. The rate constant $K_1$ is determined by measurements of the BOD at successive stations on a river. Values for $K_1$ can vary from 0.01 day$^{-1}$ to 0.3 day$^{-1}$ depending on the type of waste and the degree of stabilization (Camp, 1963). Therefore, $K_1$ values must be determined for a particular reach for the particular type of waste under consideration.

The coefficient $K_3$ for the settling and resuspension of BOD has to be determined from BOD profile measurements. Even though these are physical processes, governed by the flow conditions and particle size, it has been shown by Kranck (1974) that particulate matter from a Kraft mill effluent flocculates into different sizes depending on the particle population in the receiving media. Therefore, it is very difficult to predict the settling rate which can conceivably vary in the downstream direction. Under steady state conditions, with no sediment build-up, the amount of settling and scour should be equal. However, the material settling and that being scour may not have the same BOD concentration. Because of these factors, the coefficient $K_3$ is usually just an average for the river reach being studied. Dobbins (1964) has indicated how $K_3$ can be calculated from the measured BOD profile, provided that $K_1$ has been determined from laboratory measurements.

### 3.3.2 Immediate chemical oxygen demand

Many types of industrial effluents contain substances which are fairly quickly oxidized by molecular oxygen. These substances such as ferrous iron, sulphite, sulphide and aldehyde all exert a load on the DO content and the amount of oxygen required for the oxidation is called the immediate chemical oxygen demand. The test for this demand is given by the American Public Health Association (1971).

### 3.3.3 Plant respiration and photosynthesis

Aquatic plants, benthic algae and phytoplankton all consume oxygen for their respiration process which takes place continuously. At the same time, if sunlight is present, the plants combine carbon dioxide and water in the process of photosynthesis to produce a simple sugar and release free molecular oxygen into the environment. In general, in the daylight hours, the amount of oxygen produced by photosynthesis exceeds the amount consumed by respiration. However, during hours of darkness, photosynthesis does not take place and plants then act solely as a sink for oxygen. Therefore, dissolved oxygen levels tend to be highest in mid-afternoon and lowest during the early hours of the morning.

The photosynthesis production of oxygen depends on several factors including the light intensity, the optical density of the water and the quantity of plants (Edwards and Owens, 1965). Pescod (1969) conducted oxygen studies in a tropical alluvial river and found that even though environmental conditions were suitable for algal growth and a high level of photosynthesis, the turbidity reduced light transmission so that the actual oxygen production was very low. Nevertheless, the processes of photosynthesis and respiration can be very important factors and should be included in the DO balance.

The photosynthesis and respiration rate can be measured in one of two ways. In the light and dark bottle technique, samples of water are placed in opaque and transparent bottles and suspended in the stream for a period of time. From the initial sample, DO concentration and the final concentrations in the bottles, the photosynthesis and respiration rates can be calculated. From a DO balance of the reach and assuming reaeration and other factors to be independent of time, the photosynthesis and respiration rates can be calculated.

### 3.3.4 Atmospheric reaeration

Reaeration is the process of absorption of atmospheric oxygen into the water. It is one of the most important factors controlling the waste assimilative capacity of a river because photosynthesis and reaeration are the only two sources of oxygen replenishment, and photosynthesis can take place only when there is sunlight.

It can be shown from molecular theory that the water surface which is in contact with the atmosphere becomes saturated with oxygen in a very short time. However, once the surface is saturated the dissolved oxygen has to find its way into the body of the fluid before more oxygen can be absorbed. In the region very close to the surface, viscosity dominates and
oxygen is transferred into the liquid below it by molecular diffusion which as we know, is a very slow process. Below this surface region oxygen is quickly dispersed throughout the rest of the depth by turbulent diffusion which occurs in most natural flows. Therefore, the dissolved oxygen concentration throughout most of the flow is generally quite uniform. Although, strictly speaking, fluid elements are continuously moving in and out of the surface region, in the time average sense one can consider that there is a very thin region close to the surface in which the DO concentration drops very sharply from saturation to that for the bulk of the flow. This region is termed the turbulent surface film or oxygen boundary layer by Holley (1973) and is in many respects analogous to the viscous sublayer in the flow over a solid boundary.

It is generally agreed that the rate of oxygen absorption is proportional to the oxygen deficit, i.e. the difference between the saturation concentration and the actual DO concentration. The proportionality constant is called the reaeration coefficient $K_2$. Therefore, the source term for the right side of equation (3.23) representing reaeration is $K_2(C_s - C)$ where $C_s$ is the saturation DO concentration. $C_s$ is a function of temperature and may be calculated approximately from the following empirical equation:

$$C_s = 14.541233 - 0.3928026T + 0.00732326T^2 - 0.00006629T^3$$  \hspace{1cm} (3.25)

where $T$ is stream temperature (°C). The reaeration coefficient $K_2$ is naturally not a constant for all flows. It depends on many variables including channel slope, bottom roughness, flow velocity and wind conditions. All these factors affect the turbulence near the water surface which, as we have seen, controls the rate of oxygen absorption.

Numerous studies have been made and many prediction equations have been presented for the reaeration coefficient $K_2$. These equations are either strictly empirical, based on experimental reaeration rates, or are derived from some conceptual model of the process with coefficients which also have to be determined experimentally. The proposed models and prediction equations have been reviewed by Lau (1972) and by Bennett and Rathbun (1971). Many of the empirical prediction equations appeared to give a good fit only to the few sets of data from which they were derived. The conceptual models suffer from the fact that they all contain two or more coefficients which again depend on various flow variables and are difficult to evaluate. Reliable experimental data, especially field data, are very difficult to obtain because many factors affect the DO balance and it is difficult to account correctly for all of them. Lau (1975) found from his laboratory experiments that the dimensionless variable $K_2h/u$ is a function of both the friction factor $f$ and the Reynolds number $uh/v$ where $v$ is the kinematic viscosity of water. This shows that simple prediction equations which try to relate $K_2$ to $u$ and $h$ cannot, in general be satisfactory. Wilson and McLeod (1974) tested sixteen prediction equations against published data covering a wide range of hydraulic variables and found that even the ones with the best correlations gave very unreliable prediction rates. They suggested that the poor performance of these equations may be attributable to the omission of one or more significant variables. Of the equations tested, those of Dobbins (1964) and of Parkhurst and Pomeroy (1972) gave the best fit to all the data investigated. The latter equation is the simpler of the two and is given in terms of the mass transfer coefficient $K_L$ at 20°C. The equation is:

$$K_{L20°C} = 0.96 \left( 1 + 0.17 F_r^2 \right) (S\tilde{u})^{0.375}$$  \hspace{1cm} (3.26)

where $K_L = K_2h$ (m per hr); $F_r = Froude number (\tilde{u}/\sqrt{gh})$; $S$ = slope of energy line; $\tilde{u}$ = flow velocity (m per sec); $h$ = flow depth (m).

It is clear that not one of the existing equations can satisfactorily predict the reaeration rate over a wide range of flow conditions. Based on the results of Lau (1975) it appears that the variation of $K_2$ cannot be covered by a single equation. Therefore, the user must exercise judgement in selecting an equation for $K_2$.

3.4 PREDICTION OF WATER QUALITY CONDITIONS

So far, in the last two sections the equations governing the transport of both the conservative and non-conservative substances have been formulated and some of the commonly made simplifications of these equations considered. In this section dealing with the prediction of water quality under steady state flow conditions, some of the available solutions to these equations will be reviewed.

3.4.1 Longitudinal spreading of a conservative pollutant

Consider the longitudinal spreading of a slug of a conservative pollutant injected instantaneously in a natural stream. After the convective period, the transport of this slug of pollutant can be described by the cross-sectional average mass balance equation (i.e. equation (3.10). The
solution of this equation can be expressed as:
\[
\bar{C}(x; t) = \frac{M}{A} \frac{1}{4\pi Et} \exp \left[ -\frac{(x-\bar{u}t)^2}{4Et} \right]
\]
(3.27)
where M is the mass of the pollutants and A is the cross-sectional area of the channel.

In order to use equation (3.27) to predict the concentration distribution of the pollutant and consequently the quality of water, the dispersion coefficient E has to be known.

The dispersion coefficient E can be evaluated from tracer experiments or derived from the theoretical equations, Fisher (1968).

3.4.2 Longitudinal spreading of a non-conservative pollutant

Consider the longitudinal spreading of a slug of a non-conservative pollutant injected instantaneously in a straight channel. The pollutant is assumed to undergo a first order reaction so that the rate of removal can be represented by kC where k is a reaction rate constant. In this case the transport of the pollutant beyond the 'convective period' can be represented by the one-dimensional dispersion equation with a sink term which is:
\[
\frac{\partial C}{\partial t} + \bar{u} \frac{\partial C}{\partial x} = E \frac{\partial^2 C}{\partial x^2} - kC
\]
(3.28)
A solution of equation (3.28) is:
\[
\bar{C}(x; t) = \frac{M}{A} \frac{1}{4\pi Et} \exp \left[ -\frac{(x-\bar{u}t)^2}{4Et} + 4Ekt \right]
\]
(3.29)
where M is the mass of the non-conservative pollutant introduced into the stream.

The value of the longitudinal dispersion coefficient E can be evaluated in the same manner as in the previous case.

3.4.3 Oxygen balance

The solution of the DO concentration is a little more complicated because of the BOD term and analytical solutions are obtained only for the one-dimensional case, considering the whole cross-section of the river to be well mixed. As an example, Dobbins (1964) considered removal of BOD by sedimentation or absorption, the addition of BOD along the reach as well as the BOD reaction rate. The BOD profile equation is:
\[
\frac{dL}{dx} = \frac{E}{dx^2} (K_1 + K_3)L + L_a
\]
(3.30)
where \(L_a\) is the rate of addition of BOD along the reach and \(K_3\) is the rate of sedimentation or absorption.

The solution is given by:
\[
L = L_o \exp(mx) + \frac{L_a}{K_1 + K_3} (1-\exp(mx))
\]
(3.31)
where:
\[
m = \frac{\bar{u}x - \frac{1}{2}(\bar{u}x^2 + 4(K_1 + K_3)E)}{2E}
\]
and, \(L_o = L\) at \(x = 0\). This value for L is then used in the DO balance equation given by:
\[
\bar{u}x \frac{dC}{dx} = E \frac{d^2C}{dx^2} + K_2 (C_s - C) - K_1 L - D_B
\]
(3.32)
where \(D_B\) stands for the net effect of plant photosynthesis and respiration and benthic demand.

The solution, subject to the boundary conditions that \(C = C_0\) at \(x = 0\) and equilibrium between atmospheric reaeration and photosynthesis and benthic demand at \(x = \infty\), is given by:
\[
(C_s-C) = \frac{K_1(L_o-L_a)}{K_1+K_3} \frac{\exp(mx) - \exp(rx)}{K_2 - (K_1 + K_3)} + (C_s-C_0)\exp(rx) + \frac{D_B}{K_2} \frac{L_a}{K_2(K_1+K_3)} (1-\exp(rx))
\]
(3.33)
where:
\[
x = \frac{\bar{u}x - (\bar{u}x^2 + rK_2E)^{1/4}}{2E}
\]
Equation (3.33) is valid for describing the cross-sectional average concentration of DO when the BOD is reasonably well mixed throughout. However, outfall discharges are generally
located at one bank or at some location in the stream and are not initially well mixed across the section. Therefore, a cross-sectional variation in BOD exists across the river and until it is well mixed there will be a cross-sectional variations in DO concentration as well. In this case the two-dimensional equations have to be solved. Holley (1973) compared the one-dimensional and two-dimensional distributions and found that, except in cases of relatively fast transverse mixing, the two-dimensional solution gave smaller minimum DO concentrations and the minimum concentrations were as low as one-fourth of those calculated by the one-dimensional solution. Thus if the stretch of river under consideration includes ecologically sensitive areas such as spawning grounds, a two-dimensional DO balance should be used.

3.4.4 Management applications of flow and water quality models

3.4.4.1 Introduction
The application of the solutions described in the previous section to many water pollution problems is often of limited value because the solutions assume constant flow conditions. The steady state water quality solutions are mainly used for long-term planning problems where general information on the likely response of the system is required (Newsome et al., 1971). However, many problems in water quality management derive from transient violations of water quality standards which are governed as much by the hydrological behaviour of a system as the discharge of pollutants at points within it. Recently, Warn and Brew (1980) have drawn attention to the limitations of steady-state models in setting consent conditions for effluent discharges and attribute these limitations to the inability of steady-state models to account for the short-term interactions between flow and quality.

In applying a model to a management problem, it is important to recognise that the model is at best an imperfect representation of 'real-world' behaviour, that model simulations and forecasts are subject to error, and that decisions based on the model will be subject to uncertainty. A model which explicitly recognises the presence of this error is termed a stochastic model; models which also represent the time variant interaction between flow and quality are termed dynamic-stochastic water quality models.

3.4.4.2 Dynamic model structure
For application to design and operational management problems, a dynamic water quality model should have the following attributes:
1. the model should be capable of using time-varying input (upstream) measures of water quality to compute time varying output (downstream) responses;
2. the model should be simple, yet characterise adequately those dynamic aspects of system behaviour of particular importance for the problem at hand;
3. the model should provide a reasonable mathematical approximation to the physico-chemical changes occurring in the river system and should be calibrated using measurements collected from the river at a sufficiently high frequency and for a sufficiently long period of time;
4. the model should account for the inevitable errors associated with laboratory analysis and sampling, as well as the uncertainty associated with imprecise knowledge of the pertinent physical, chemical and biological mechanisms.

As indicated in Figure 3.9, the function of the dynamic model is to link the water quality model with the hydrological model such that interactions are incorporated directly. The dynamic water quality models employed in several major water quality studies (for example, The Bedford Ouse Study - Whitehead et al., 1979, 1981) are based on a transportation delay/continuously stirred tank reactor (CSTR) idealisation of a river (see Figure 3.10). The mathematical formulation of this model is in terms of lumped parameter ordinary differential equations and draws upon standard elements of chemical engineering reactor analysis, e.g. Himmelblau and Bischoff (1968).

The principal advantages of this lumped-parameter model over the corresponding partial differential equation model are the following:
1. the computational effort associated with the solution of the equations is greatly reduced;
2. statistically efficient algorithms for model structure identification and parameter estimation can be readily utilized.

While the parameters of steady-state models have traditionally been estimated either from laboratory experiments or through reference to the literature, the parameters of dynamic models can be estimated directly from field data, thus ensuring that the model behaviour will be representative of processes occurring at the field scale.

3.4.4.3 Applications of dynamic water quality model
There are two principal uses for dynamic water quality models:
1. to supplement the use of steady-state models at the planning/design stage in carrying out
Figure 3.9 Integrated flow and quality model (Whitehead et al, 1981)
Figure 3.10 Continuous stirred tank representation of a river reach (Whitehead et al, 1981)
more detailed investigations of alternative system designs, where a system, in the present context, is a configuration of storage reservoirs (surface and groundwater), sewage and water treatment plants linked through a network of natural and artificial water courses. Specifically, dynamic models can be used to evaluate the efficacy (ultimately measured in terms of cost-effectiveness) of different designs in maintaining standards at critical points in a system and of exploring some of the subtle effects which can occur over time (e.g. the build-up of a pollutant in a pumped storage reservoir through continued recycling of water). A typical time unit for such models would be one day and water quantity simulation models on this time scale have been used for several years in evaluating the reliability of systems for water supply;

2. to assist in the operational control of river systems. The increasing use of automatic water quality monitors and telemetry affords considerable scope for hour-to-hour decision-making through the use of short-term forecasts of water quality. Such forecasts are particularly useful when abstracted water is to be blended with waters of different quality from other sources such as reservoirs or aquifers.

3.5 MAJOR TYPES OF BIOTIC CHANGES CAUSED BY SELECTED POLLUTANTS

3.5.1 Introduction

Aquatic ecosystems react to pollutants by changing their structure and dynamics in respect of quality and quantity. The effects of pollutants depend on type and actual condition of the aquatic ecosystems under consideration, and on the dose/effect relation between the pollutant and the organisms concerned. Generally, the ecological consequences of the inflow of pollutants can be classified as follows:

1. Acute consequences
   1.1 Inhibition of biological activity and changes in the aquatic ecosystems.
   1.2 Impoverishment of the aquatic ecosystems.
   1.3 Depopulations and fish kill.

2. Chronic effects.
   2.1 Bioaccumulation of persistent pollutants in organisms.
   2.2 Concentration and effects of pollutants in the nutrient chain, reduction in the quality and quantity of aquatic organisms, e.g. fish.

It is proposed here to consider two groups of pollutants classified according to their primary effects:

1. Substances which settle or cause turbidity; and
2. Oxygen-consuming substances.

3.5.2 Substances which settle or cause turbidity

Turbidity due to suspended matter and its sedimentation is a natural phenomenon encountered in many streams, particularly in large rivers where it occurs continuously or periodically. Only organisms adapted to such conditions live in these surface waters. Their nutrient base is, to a large extent, of allochthonous origin. Many other medium and small watercourses carry naturally clear water with turbidity occurring only temporarily during flood periods. Hence turbidity and sedimentation remain ineffective, or have only a moderate effect, mostly in the form of a strictly local deposition of fine material which permits the development of a variety of minor biotopes on the bottom of the watercourse.

Inorganic suspended sediments enter the watercourses in the form of coal particles, iron oxide or china clay, mostly from the ore and stone industry. Organic, poorly degradable substances such as those from the wood and paper industry reach the rivers mainly through waste water. The primary effect of substances causing turbidity and settling when the tractive force of the water decreases is a deterioration in light intensity. Activity and growth of aquatic plants (aquatic macrophytes, moss and algae) are inhibited even to the extent of complete elimination.

Thus, the food base of the animals dependent on these plants is also destroyed. Suspended solids quickly clog the filter apparatus of certain insect larvae (e.g. caddis larvae), which normally use their apparatus to trap organic particles for nutrition. With increasing deposition of fine material on stones, gravel and wood in the water, the food source of organisms living on solid substrate (algae, moss, snails, caddis larvae, etc.) is destroyed. This also affects the mobile organisms which depend on the substratum as a physical base, such as stone, ephemeral and caddis fly larvae, or those which graze on it, such as snails. Other mobile aquatic animals like Gammarus (a very important fish-food in the salmonid region of streams and rivers) can no longer develop and fish production will be reduced.
Fine-grained sediments, usually considered as silt deposits, are inhabited by worm-shaped organisms (sludge worms and some chironomid larvae) which are specially adapted to this biotope, provided that sediment accretion is slight and that the material itself (e.g. iron oxide sediment) is not hostile to colonization. Generally, sedimentation of fine particles destroys the diversity of the habitat. In addition to these purely physical effects of the substances causing turbidity, it must be realized that, simultaneously with these substances, other pollutants and toxic agents reach the water intensifying the effects of the suspended substances on the aquatic organisms. Detailed information of this matter is compiled by Hynes (1960).

3.5.3 Oxygen-consuming substances

Changes in the oxygen content of surface waters may have natural causes or result from human influence. Natural fluctuations of the oxygen concentration in surface waters result from the activity of autotrophic plants which produce and consume oxygen in a day-and-night rhythm.

Changes in the oxygen budget which are caused by waste water are of much graver consequence for aquatic life. Oxygen-consuming substances in waste water may be substances which react chemically with the dissolved oxygen in the water. The best-known example is water with a high content of iron II salts which are oxidized in the water by the dissolved oxygen to form iron III oxide. In this case, the effect on the oxygen content and the organisms in the water is limited to a smaller or larger part of the surface water and will persist until the oxygen loss is compensated for by surface reaeration. Iron polluted water usually is acidic and enters the watercourses from mines. Ferrous iron salts together with the acidity of the water lead to an excessive development of filamentous iron-bacteria and reduction in most of the other organisms (Klein, 1962).

The main problem is created by substances which are degraded biochemically with oxygen being consumed in the process, i.e. substances mineralized by bacterial activity. This applies equally to domestic, agricultural and industrial waste waters. Actual effects depend on quantity and concentration of the biologically degradable substances and on their dilution in the water. Physiographic conditions are also involved because part of the biochemical oxygen consumption is compensated for as a result of physical aeration as effected, for example, by turbulent flow.

Biochemical oxygen consumption in water results from the activity of heterotrophic bacteria which use organic substances for nutrition. The first change in aquatic life is, therefore, an increase in the number of bacteria and an intensification of bacterial activity. At the same time, other changes take place which, with increasing pollution, lead to ever new biological structures as, in addition to the oxygen content, a variety of other river characteristics also change. These include: turbidity and the resulting reduction in light intensity; increased food supply for filtering organisms and a rise in the concentration of mineral nutrients which stimulate the growth of algae and aquatic macrophytes. This is followed by further changes in the ecological system which manifest themselves visibly in the structure and dynamics of the aquatic biotopes. This dependence of the colonization on the effects of the biologically degradable, oxygen-consuming substances is the basis of biological water quality monitoring and assessment (cf. Guidebook on Water Quality Surveys, Unesco/WHO, 1978, and Hynes, 1960). With water pollution increasing, the following seven stages can be observed:

3.5.4 Stages of biotic change

Stage 1
Clean waters with a naturally limited supply of plant nutrients. Flow is rapid and turbulent. The water is clear and always in a state approaching oxygen saturation. The water is cool, even in summer, especially in temperate latitudes and altitudes. Colonization is sparse, or moderately dense at best. It includes long-living algae, in particular red and blue-green algae (e.g. Batrachospermum, Hildenbrandia, Lemanea, Calothrix parietina and Phormidium inundatum). Various diatoms and moss species, as well as animals such as Planaria alpina, and insects (stone fly and caddis fly larvae) may be found. The larvae have external gills and their cycle of development may last from one to several years.

The nutritional basis of the animals consists primarily of washed-in plant material that is rich in cellulose content. These waters are spawning grounds for salmonids whose survival depends particularly on the availability of an adequate oxygen supply.

Stage 2
The same basic characteristics as those of stage 1, i.e. rapid and turbulent flow and cool water in summer. Slightly increased inflow of inorganic nutrients and small quantities of organic,
biologically degradable substances leads to denser colonization and greater species variety. This occurs, e.g. with the inflow of small volumes of domestic sewage which do not cause significant reductions in the oxygen content. At this low concentration, the influence of putrescible organic substances manifests itself after a short distance of self-purification primarily as a fertilizing effect. The presence of filamentous green algae (e.g. Ulothrix zonata) and of certain diatoms (e.g. Meridion circulare) in the community is particularly evident. Insect larvae with external gills but small gill surfaces, such as Amphinemura, Perla marginata, some caddis, stone and ephemeral fly larvae, predominate. The development cycle of the animals is usually one year. The nutritional base of the animals consists of autochthonous material (algae and moss).

These reaches are also of importance as spawning grounds for salmonids and other fish requiring clean, well oxygenated water.

Stage 3
The current in these streams ranges from rapid to slow. In summer, the water may be either cold or warm. The degree of pollution by oxygen-consuming substances occurs under natural conditions if the drainage area offers a rich nutrient supply.

The higher temperatures cause more intensive biological activity. Moderate pollution by organic, biologically degradable substances is quickly reduced by the self-purification process. The degradation products have a stimulating effect on plant growth so that locally limited deposition of silt with autochthonous organic substances may occur at certain points. Oxygen consumption during biological degradation is low but there are marked diurnal variations in oxygen content caused by photosynthesis and plant respiration. Aquatic plants may cover large areas of the water surface, at least during the main growing period. The biomass of the algae is considerable; in slow-flowing streams, the water may be turbid as a result of the mass development of planktonic algae.

All important groups of fresh water algae occur in this zone. The outstanding characteristics, however, are the great variety of species, and in temperate climates there is a change of species during the year. Spring with cool water and low light intensity: diatoms; summer with warm water and high light intensity: green algae; autumn with relatively warm water and low light intensity: decrease in the number of green algae and renewed increase in the diatoms; winter with cool water and low light intensity: decrease in the number of all autotrophic plants.

The animals also reach maximum species variety in this zone, and it is characteristic that many kinds of snails, small crustaceans and insect larvae reach high population densities. Most insect larvae have exterior gills which offer a large surface area for gaseous exchange. Two or more generations per year may be produced. Due to the rich supply of vegetable food (algae, macrophytes), grazers and filter-feeders predominate. Of the predatory animals present, most are insects and their larvae, in particular dragon-flies.

As a result of the rich stock of fish food organisms and the good oxygen supply, this zone is highly productive and used for spawning by salmonids and cyprinids.

Stage 4
Continued increase in the pollution by oxygen-consuming substances leads to a condition of 'critical pollution'. It is characterized by a wide amplitude between maximum and minimum oxygen concentration. These waters usually support a dense growth of algae and aquatic macrophytes. Among them, and on the solid substratum, live many filtering organisms, detritus eaters and grazers. Predacious species (leeches, beetles, water bugs) will occur. Stone fly larvae and other animals adapted to a permanently high oxygen content of the water are lacking. Generally speaking, there is less species diversity than in Stage 3. A few forms, on the other hand, show a tendency towards mass development. As a rule, the cycle of development is short, many species producing several generations in a year.

The rich supply of organic substances leads to intensive bacterial activity and the high rate of self-purification is associated with a high rate of oxygen consumption. Mineralization of organic substances leads, at the same time, to an increase in the nutrient supply so that the aforementioned mass development of algae and macrophytes can take place. Heavy oxygen supersaturation in daytime alternates with a marked reduction in oxygen concentration at night. Under unfavourable conditions, the oxygen content of the water may drop, especially in the early hours of the morning, to a point where fish may be killed even though surface waters of this stage often are high yield fish waters.

To cope with the wide amplitude of oxygen concentrations many of the insect species (e.g. beetles and predatory bugs) have organs for extracting oxygen from the atmosphere and are air breathers. Other characteristics are the occurrence of many ciliates which feed primarily on bacteria, and the development, already visible with the naked eye, of small colonies of the so-called sewage fungus (Sphaerotilus natans).
Stage 5
Further increase in pollution by waste water containing organic, biologically degradable substances usually results in milky turbidity. Bacterial activity is so intensive that low oxygen concentrations predominate. Aquatic macrophytes and algae may be present in sufficient numbers to influence the oxygen budget, but they do not visually dominate. Extensive colonies of sewage fungus can be observed with associated populations of bacteria and ciliate protozoans. Only a few species of animals can tolerate the low oxygen content but these, as a rule, have a high population density. They are animals of little sensitivity to oxygen scarcity, such as isopods, leeches and sponges. Blood worms (Chironomus thummi) and sludge worms (Tubifex sp.) whose blood pigment allows them to store oxygen and to make optimum use of low oxygen concentrations, settle on the digested sludge deposited in this zone.

The larger animals are mostly predatory (eg leeches and diving beetles).
At this level of extreme adaptation, it is not only the oxygen concentration which plays a dominating role in the selection of the species. A complex of diversified effects is already apparent. With decreasing oxygen content, the concentration of intermediate products of biological degradation, such as ammonium, nitrite and hydrogen sulphide, increases. These substances can either consume oxygen by further oxidation or they can have toxic effects, at least in higher concentrations (hydrogen sulphide), i.e. non-toxic ammonium (NH₄) changes into toxic ammonia (NH₃) at high pH values.
While fish life is still possible under these conditions, the yield is poor and periodic fish kills occur.

Stage 6
Heavier pollution leads to a high degree of turbidity and to increased concentrations of noxious metabolic products of microbial degradation, with the result that the living conditions of macro-organisms are severely limited. Autotrophic plants do not occur and the biogenous oxygen supply is lacking. The bottom of the river is covered by digested sludge on which only a few species are living; these, however, occur frequently in large numbers. They feed almost exclusively on detritus. Tolerance to low oxygen concentrations is ensured by possession of haemoglobin (eg blood worms, sludge worms), or else oxygen is taken from the atmosphere, eg the larvae of Eristalis tenax which have a breathing tube that can be expanded to a length of 15 cm and by means of which they can reach the air above the water surface. The predacious macro-organisms rather frequently encountered in the preceding stage now occur only sporadically.
Fish cannot develop under these conditions, at best, they can briefly stay at favourable points.

Stage 7
With even heavier excessive pollution by oxygen-consuming waste water constituents, putrefaction processes prevail. The water is very turbid and usually smells clearly of hydrogen sulphide. Life is reduced to bacteria, colourless flagellates and ciliates feeding on bacteria. A typical feature is the mass occurence of sulphur bacteria which can assimilate sulphur compounds or hydrogen sulphide. Autotrophic organisms and fish are not present.

REFERENCES


4 Characteristics of lakes and reservoirs
4.1 INTRODUCTION

To understand the effects of pollutants on lakes and reservoirs, some account has to be taken of their general characteristics and properties. In the space permitted here only the basic features can be described; for detailed information the reader is referred to more extensive texts (Hutchinson 1957, Ruttner 1962, Dussart 1966, Lowe McConnell 1966, Golterman 1975).

Lakes and reservoirs are typically standing waters, the former naturally occurring and the latter man-made. They exhibit a vast range of surface areas, volumes, depths and water retention times. Many lakes have enormously long retention times, often measured in hundreds of years while most reservoirs have retention times considerably less. Many reservoirs with very short retention times possess some of the characteristics of rivers, e.g. horizontal gradients in chemical parameters and water currents. Orlob (1969) devised an index to describe how close a body of water is to the riverine or lacustrine situation. The higher the value of the index the closer the body of water is to behaving like a river and not exhibiting thermal stratification.

\[ Fr = 320 \times \frac{L}{d} \times \frac{Q}{V} \]

where \( Fr \) = Stratification index
\( L \) = Length of water body (m)
\( d \) = Average depth (m)
\( Q \) = Average discharge from water body (m³/s)
\( V \) = Volume (m³)

Three examples of the index are given in Table 4.1.

<table>
<thead>
<tr>
<th>Reservoir</th>
<th>Length (m)</th>
<th>Mean depth (m)</th>
<th>Discharge to volume ratio (sec⁻¹)</th>
<th>Fr</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hungry Horse</td>
<td>4.7 x 10⁴</td>
<td>70</td>
<td>1.2 x 10⁻⁸</td>
<td>0.0026</td>
<td>Stratified</td>
</tr>
<tr>
<td>Lake Roosevelt</td>
<td>2.0 x 10⁵</td>
<td>70</td>
<td>5.0 x 10⁻⁷</td>
<td>0.46</td>
<td>Weakly stratified</td>
</tr>
<tr>
<td>Wells</td>
<td>4.6 x 10⁴</td>
<td>26</td>
<td>6.7 x 10⁻⁶</td>
<td>3.8</td>
<td>Completely mixed</td>
</tr>
</tbody>
</table>

(1) Montana, USA
(2) Columbia River, USA

Because of climatic conditions, evaporative losses can have an intensive influence on shallow lakes particularly so in arid or semi-arid regions. If there are no natural outflows large changes in water level can occur. In contrast, man-made reservoirs usually have a
regulated outflow and depending on their use and the climatic conditions may experience no
drawdown at all or a regular draw-down/refill regime. This leads to differences in the
behaviour of lakes and reservoirs, referred to later.

There are two main types of reservoirs:
1. Those created by damming a river and flooding its valley.
2. Those created by constructing an artificial basin and filled by pumping or gravitation.
Both types of reservoir can be used for several purposes, e.g.
1. Reservoirs primarily intended to store water in times of plenty for use for drinking
or irrigation during times of shortage. These reservoirs may be operated on a 'put and take'
basis, i.e. being maintained at top water level by matching demand for supply with abstraction
from the raw source or be filled during the wet season and drawn down during the dry season.
2. Flood control reservoirs. These usually have a controlled falling water level during
the major part of the year so that there is sufficient storage capacity to accommodate floods.
In extreme cases, the reservoir may be filled in a few days.
3. Pumped storage reservoirs are used for power generation and may be filled and emptied
on a 24-hour cycle.
4. Power station reservoirs are usually kept at top water level so as to ensure constant
power generation or cooling.
5. Reservoirs for recreation usually require a constant water level.

4.2 PHYSICAL AND HYDRAULIC CHARACTERISTICS

4.2.1 Thermal stratification

It is convenient to begin by considering a lake in a temperate region in spring when the
entire body of water is at a uniform temperature, at or slightly above 4°C, the temperature of
maximum density. Increasing solar radiation warms the surface layers and currents set up by
wind stress on the water surface generate turbulent motion leading to mixing and downward
transport of heat. In all lakes of sufficient depth, heating in the spring from a low
temperature, the water tends to become divided into an upper region of more or less uniformly
warm, circulating turbulent and illuminated water termed the *epilimnion*, and a deep, cold
relatively undisturbed and dark region termed the *hypolimnion*. The two layers are separated
by a relatively narrow layer of water of rapidly decreasing temperature with depth termed the
*metalimnion*. This development reaches its peak in summer and is called the summer stratifica-
tion.

During the autumn there is no gain in heat and the lake begins to cool. There is a
steady increase in thickness of the epilimnion as its temperature falls and initially the
deep layers of the hypolimnion may be hardly disturbed. As the cooling proceeds, the
epilimnion finally includes the whole lake which is then in full autumnal circulation. This
event is called the autumnal or fall overturn. The lake may continue to circulate throughout
the winter at low temperatures but if prolonged ice cover develops, it is possible to develop
an inverse stratification where colder, less dense water may overlie water at 4°C. This
situation may prevail until spring or ice-melt when the water is once again isothermal and
freely circulating. This event is called the spring overturn. The whole cycle is illustrated
diagrammatically in Figure 4.1.

Hutchinson and Loeffler (1956) have classified lakes according to their thermal behaviour
and recognised six main categories:
1. Cold Monomictic - usually polar, sub-polar or high mountain lakes with full circulation
only in summer usually being free from ice for only a short period.
2. Dimictic - temperate lakes with full circulation in spring and autumn, stratified in
summer. (This type is described in detail above).
3. Warm Monomictic - sub-tropical lakes, stratified for most of the year, full circulation
in early winter.
4. Oligomictic - small or very deep lakes where a very small temperature difference between
top and bottom surfaces maintains stable stratification. The lake circulates only at
rare, irregular intervals when abnormal cold spells occur.
5. Warm Polymictic - tropical lakes receiving solar radiation during the day but back-
radiating at night allowing nocturnal mixing.
6. Cold Polymictic - lakes at great altitudes, no permanent stratification develops, mixing
at night.
Figure 4.1 Annual thermal cycle of a temperate, dimictic lake
4.2.2  Water movement

In general, current systems in lakes are of two kinds. The first is periodic and generally due
to the disturbance of the lake by wind forces or atmospheric pressure causing an oscillation
of the lake or part of it. If a wind that has piled up water at one end of the lake suddenly
dies down, a current will flow momentarily restoring the lake to its former level. The
current will not have lost its energy and will continue, piling up water at the opposite
end of the lake. This current or seiche constitutes an oscillation about a nodal line. As
with other oscillating systems there is a tendency for harmonics to form and seiches may
be multi-nodal.

The second kind of current is non-periodic, generated by external forces namely the
influent-effluent system of the lake, unequal heating, the entry of dissolved material from
the sediments and variations in atmospheric pressure and winds. Special problems arise in
very large lakes where the Coriolis force (an inertial current imposed by the earth's
rotation) is of significant influence. One of the effects is the existence of a vertical
'thermal bar' between warm, inshore waters and cold, offshore waters. For details see
Murthy and Blanton (1975) and Rodgers (1968). An example of the effect of the influent-
effluent system of the lake has been described by Hutchinson (1957) for Lake Constance where
the course of the River Rhine through the lake has been tracked (see Figure 4.2).

Special problems with the behaviour of incoming water may also arise in reservoirs
especially but not exclusively in the narrow, elongated river impoundment type. If there
is a pronounced temperature (and density) difference between the influent and the reservoir,
the incoming water may plume across the surface or flow along the bottom. In either case
short-circuiting of the system will occur and many of the benefits of storage will be lost.
Uhlmann (1975) gives a schematic view of the basic flow and stratification patterns of
reservoir water (Figure 4.3). Flow phenomena influence retention time so there may be large
differences between the theoretical and actual retention times in reservoirs.

Further problems due to temperature may arise depending on the position of the reservoir
outlet. The demand placed on a reservoir to provide water affects the water level, the
dynamics of the impounded water and the conditions downstream of the reservoir. If the
reservoir outlet is close to the surface, the river downstream will receive warm water in
summer and cold in winter. From deep stratified reservoirs with outlets at the bottom,
cold hypolimnetic water may be released continuously thus providing a habitat downstream for
cold water adapted fish.

In pumped storage reservoirs, temperature regime and stratification depend to a
considerable degree on the mode of operation. Conditions in shallow reservoirs for drinking
water supply correspond to those in natural lakes. In deep pumped storage reservoirs used
for power generation, there may be no stratification because of the rapid flow through of
water.

4.3  CHEMICAL AND BIOLOGICAL CHARACTERISTICS

Information on the general chemical composition of lakes and reservoirs can be obtained from
the basic texts referred to in the introduction to this chapter. Although many elements
have been shown to be essential for aquatic life this chapter will concentrate on phosphorus
and nitrogen as the main chemical factors influencing primary production. The complexity
of the cycling of elements within aquatic ecosystems can be seen in Figure 4.4.

Thermal stratification in standing water-bodies strongly influences the biology and
chemistry. The epilimnion is usually well illuminated (euphotic) and, given sufficient
nutrients such as nitrogen and phosphorus, planktonic algae develop and grow. The epilimnion
can then also be said to be trophogenic. Where extensive growths of algae occur there may
be increases in the pH as the carbon dioxide and bicarbonate ions in solution are converted
to organic matter. Diurnal fluctuations in dissolved oxygen concentration may also occur.
Most of the organic matter produced in the epilimnion falls into the dark, cool hypolimnion.
Consumption and conversion of this rain of particulate matter is effected by animals and
bacteria. This process can be likened to the self-purification system occurring in rivers.
However, in the hypolimnion the supply of oxygen is strictly limited and it is possible for
the decomposition processes to use up the available oxygen, producing anaerobic conditions.
Unless man intervenes to prevent or ameliorate this situation, there will be no replenishment
until next circulation or turnover.

The intensity of photosynthetic activity varies enormously as can be seen in Figure 4.5
taken from Talling (1965). An example of intense deoxygenation occurring in the hypolimnion
is shown in Figure 4.6. The graphs show that for approximately 7 months the hypolimnion was
Figure 4.2 Ordinary disposition of surface currents in Lake Constance. Dotted line indicates typical course of the Rhine (from Hutchinson, 1957)
Figure 4.3 Schematic diagram of flow and stratification patterns in a reservoir (from Uhlmann, 1975)
Figure 4.4 Nitrogen and phosphorus cycles in aerobic aquatic ecosystems (from Watanabe, 1978)
Figure 4.5 Comparative measures for the seasonal diatom maxima in Windermere (14th May 1959) and Lake Victoria (3rd August 1961) of depth profiles, of population density, depth profiles of photosynthetic activity and of derived estimates of photosynthetic activity per unit area (from Talling, 1965)
Figure 4.6 Seasonal changes in water temperature and dissolved oxygen in Lake Kariba 1960-1963. The H$_2$S zone in 4.6b indicates oxygen less than 1 mg O$_2$/litre (from Lowe-McConnell, 1966)
almost entirely anaerobic, allowing poisonous hydrogen sulphide to be formed.

This interaction between lake morphology, hydrology, chemistry and biological activity permits the construction of a lake classification (or typology) scheme. It is an accepted fact that this is essentially a subjective classification without clear definitions of borderlines of change.

In this system we have, on one hand, oligotrophic lakes which are usually clear and deep with low concentrations of nutrients and little plankton activity. Bioproduction and other bioprocesses have little influence on ionic equilibrium and chemical concentrations. At the end of the stratification period the hypolimnion will still have well oxygenated conditions (70% of saturation).

On the other hand, eutrophic lakes may be turbid because of suspended plankton with secchi disc visibility of less than 2m deep. High concentrations of nutrients stimulate intense activity of phytoplankton producing supersaturation of oxygen in the epilimnion. At the end of the stratification period the oxygen concentration in the hypolimnion may vary from 0-30% of saturation.

Excessively eutrophic lakes have been described variously as polytrophic, hyper-eutrophic or hypertrophic. Some characteristics of hypertrophic lakes are the presence of hydrogen sulphide in the totally anaerobic hypolimnion and massive developments or blooms of algae at the surface.

The intermediate phase between oligotrophic and eutrophic may be termed mesotrophic.

The basis of data collected from a large number of lakes worldwide, Vollenweider, 1968, 1976) has produced a series of correlations aimed at defining and predicting the trophic state of a body of water within certain limits or probability. One such prediction is shown in figure 4.7 which relates the average chlorophyll concentration in a body of water to the phosphorus load imposed upon it taking due account of retention time and mean depth.

One of the outcomes of a worldwide research programme coordinated by OECD is that definite limits have now been proposed for the trophic categories described above. The limits are based on average phosphorus concentration, mean and maximum chlorophyll concentrations and mean and maximum secchi disc transparencies (Vollenweider & Kerebes, in press).

Any lake or reservoir can be divided into several zones or biotopes which differ markedly in their biological characteristics.

The littoral zone (Figure 4.8) is shallow and illuminated and characteristically inhabited by aquatic plants if the substratum permits. Obviously sand, clay or silt margins will support a richer flora than igneous rock. The littoral is often richly productive being inhabited by snails, insect larvae and worms. It is an important feeding ground for water birds and an important spawning ground for fish. In a shallow lake or reservoir it may play a dominant role.

Below the littoral lies the benthal zone which may well be illuminated and productive in its upper part and dark and non-productive in its lower. It is an important habitat for algae and bacteria and protozoa, worms and insect larvae are often to be found in abundance.

The main water mass is termed the pelagial and in most lakes is the dominating biotope. The upper, illuminated *trophogenic zone* is the habitat of phytoplankton - microscopic algae adapted to a floating life. Feeding upon them, or each other or any suspended particle matter are the floating animals or zooplankton. The level of primary productivity decreases with depth until a certain point is reached where destructive processes equal the productive processes and this may be termed the *compensation depth*. Below this is the *tropholytic zone* where the organic matter produced above is decomposed or digested.

Some of the characteristics of lake biotopes may be different or absent in reservoirs because of their mode of construction or operation.

Normally, reservoirs with large fluctuations in water level and steep banks offer unfavourable conditions for the development of a littoral community. This may be advantageous in reservoirs supplying drinking water but disadvantageous in reservoirs intended for recreational use such as fishing.

4.4 DISPERSION OF POLLUTANTS

The circulation and water movements in natural bodies of water such as lakes are generally very complex turbulent motions. Superimposed on the mean flow circulation patterns are eddy-like motions of varying intensities and scales. These eddy-like motions exist in both horizontal and vertical directions; the size of horizontal eddies, however, is much greater than the size of vertical eddies because the horizontal dimensions of lakes are greater than their depth.

When a 'batch' of pollutant is released into a turbulent lake-current, it is subject to two important physical effects of water movement: transport and diffusion. The former is
Figure 4.7 Prediction of average chlorophyll and trophic character of lakes relative to phosphorus loading characteristics (from Vollonweider, 1976)
Figure 4.8 Lake or reservoir biotypes
bulk bodily movement of a parcel of pollutant due to lake mean currents, whereas the latter is the spreading of the pollutant parcel as a consequence of turbulence and current shear naturally associated with the lake currents. While both these processes are irregular and random, both contribute to the effective dilution. The emphasis in this discussion will centre around how lake environmental processes contribute to the transport and dispersion of pollutants discharged into the lakes.

The concentration field $C(x, t)$ within a diffusing pollutant 'cloud' or a plume in the lake environment is a random variable; it varies irregularly in both space and time. Iso-concentration contours mapped from an individual diffusing fluorescent dye patch usually have irregular shapes and the diffusing dye patch grows in a highly complex manner. An example of iso-concentration maps from a carefully conducted dye patch diffusion experiment in Lake Ontario during the International Field Year on the Great Lakes is shown in Figure 4.9. Cross-plume concentration profiles in the wake of a continuously maintained pollutant plume are similarly irregular and random (Figure 4.10). It is customary to remove the randomness from the observed concentration distributions by appropriate averaging. For example, in the case of dye patch concentration distributions shown in Figure 4.9, it is customary to convert it into an equivalent radial symmetric distribution to remove the randomness from the observed concentration distributions. In the case of continuous plumes one may overlap the individual concentration profiles and then the averaging is carried out. In this manner, one obtains some sort of a mean concentration distribution. In most investigations of transport and dispersion processes in the lake environment, one is generally concerned with an averaged concentration field although the fluctuations of concentrations are quite often comparable in magnitude to the local mean (Murthy and Csanady, 1973). Due to space limitations, we will not discuss the fluctuation problem here, instead we refer the reader to an excellent review of the topic by Csanady (1973).

Many pollutants retain their chemical composition in the lake environment for long periods of time and as discussed in previous chapters such substances are called 'conservative'. When a given quantity of conservative pollutant is introduced into the lake, it gradually spreads both in horizontal and vertical directions. Because the 'total mass' is conserved, the pollutant concentration at the centre of the patch must decrease in time and this dilution is described as mixing of the pollutant with the ambient fluid. The two important characteristics of diffusing plume or patch are the spread defined in terms of the standard deviations in the three dimensions and the maximum concentration. These two dispersion characteristics are related through the conservation of mass. Therefore, in predicting the dispersion of a pollutant patch or plume, we need only to focus on the rate of growth of the diffusing plume or patch under lake environmental conditions. Empirical diffusion characteristics to be presented later are indeed related to this important parameter.

There are several conceptual models of turbulent diffusion which are directly applicable to describe the transport and dispersion of pollutants discharged into a lake, and, in general form, these are similar to the differential equation models described for dispersion, in river systems but extended to the three dimensional case. We will not delve into the detailed discussion of these models as applied to the lake situation here but the reader is referred to excellent reviews by Okubo, 1962 and Csanady, 1973.

4.4.1 Diffusion processes

Turbulent fluctuations of current velocity may be visualized as comprising many eddies of different sizes. These eddies are radically different in the horizontal direction as compared to the vertical direction. When a diffusing patch or plume of pollutant comes under the influence of an eddy, it simply advects with the eddy and does disperse appreciably. However, eddies which are comparable in size to that of the patch lead to effective dispersion. This process usually referred to as 'eddy diffusion' goes on as long as there are eddies of the required size present in the lake currents.

Dispersion of a pollutant cloud in the lake environment is caused by yet another process known as 'shear diffusion' due to the vertical and horizontal shears associated with the lake currents. An interaction of current shear and eddy diffusion results in an apparent dispersion of pollutant clouds. The effect of current shear is particularly important in large lakes where diffusing clouds of pollutant often become large enough for their mean velocities to be significantly different from the leading edge to the trailing edge. Clearly, such velocity differences distort the shape of the cloud. The immediate effect of such distortion is a sharpening of the concentration differences existing in the cloud. Turbulent mixing across these sharpened concentration gradients increases the flux and produces a cloud growth rate which can be much in excess of what it would be without the
QUASI-SYNOPIC HORIZONTAL DISTRIBUTION OF DYE CONCENTRATION (ppb).

DYE RELEASE

DYE PATCH CENTRE OF GRAVITY
= DYE RELEASED 1400Z 27/6/72
- 7 HRS. AFTER RELEASE
- 26 HRS.
- 53 HRS.

CURRENTS
SCALE: RADIUS = 10 CM/SEC.

Figure 4.9 Distribution of tracer in a lake (after Murthy, 1976)
Figure 4.10 Tracer concentrations across a plume (from dye experiments conducted on Lake Ontario, see Murthy, 1976)
distortions. These dispersion processes are complicated in the presence of a thermocline, and the density stratification associated with the thermocline forms a 'diffusion floor'. A 'diffusion floor', by definition, is an impenetrable barrier below which no trace of pollutant could be detected, in spite of very high concentrations immediately above. Diffusion floors occur quite frequently during spring warming phase and summer heating of the annual thermal cycle in lakes. In the presence of a 'diffusion floor' downward diffusion of pollutant is inhibited because vertical turbulence is suppressed by stability. Another feature of the thermocline is that large current shears are generated when the thermal stratification of lake water is strongly stable, enhancing horizontal dispersion in the upper mixed layer.

As discussed briefly above, several processes contribute simultaneously to the transport and dispersion of pollutants in the lake environment. Where such processes can be quantified, it may be desirable to consider their contributions to transport and dispersion separately. However, in practice this is rather difficult and it is only possible to determine an effective coefficient of diffusion representing the overall effect, which one can attempt to relate empirically to the physical processes of the lake environment. We will present here some diffusion characteristics which are fundamental for prediction of dispersion of pollutants in the lake environment.

Horizontal scales of motion in lakes are much greater than the vertical scales and, therefore, their effect on diffusion can be considered separately. This idea has been explored by a number of investigators. In this approach, it is assumed that the introduced pollutant is subject to horizontal mixing within a sufficiently thin homogeneous layer so that all variations in both concentration and velocity may be neglected. However, the importance of vertical diffusion cannot be neglected since the combined action of vertical shear in the horizontal mean current and vertical diffusion may produce considerable effective horizontal diffusion.

A diffusion characteristic of considerable importance in the prediction of pollutant concentrations is the horizontal eddy diffusivity versus the length scale of diffusion. Based on a large number of fluorescent dye tracer diffusion experiments under field conditions, these characteristics have been recently constructed empirically by Murthy (1976) following Stommell (1949) and Okubo (1971). Figures 4.11 and 4.12 show such characteristics. The following regression equations were obtained between the dispersion coefficient (cm$^2$ sec$^{-1}$) and the length scale of diffusion (cm):

\[ K_x = 1.3 \times 10^{-2} L_x^{1.3} \]
\[ K_y = 2.4 \times 10^{-3} L_x^{1.3} \]
\[ K_y = 1.8 \times 10^{-3} L_y^{1.4} \]
\[ K_y = 2.2 \times 10^{-4} L_y^{1.4} \]  

(4.1)

As pointed out earlier, a wide spectrum of horizontal motions exists in the lakes. Thus, the horizontal eddy diffusivity usually increases with the length scale. As expected, however, the hypolimnion values are one to two orders of magnitude smaller than those of the epilimnion values. A physically acceptable explanation for the smaller values at greater depths could be that the available turbulent energy for mixing decreases with depth. An important conclusion that can be drawn from these diffusion characteristics is that the eddy diffusivity grows faster than given by the Fickian model and somewhat slower than in the 'inertial subrange'. Although the diffusion characteristics do not support the familiar turbulent diffusion theories, they could be viewed as purely statistical since they have been constructed from experimental data obtained in widely varying environmental conditions. The dependence of the horizontal eddy diffusivity on the length scale provides a useful guideline for modelling practical dispersion problems.

In contrast to the horizontal diffusion, the process of vertical diffusion is controlled primarily by small-scale motions characteristic of stably stratified water. Wind mixing plays an important role in providing turbulent energy to the epilimnion and possibly down to the thermocline. Unlike the horizontal eddy diffusivity, the vertical diffusivity does not depend uniquely on the scale of diffusion. Efforts have been directed to relate the vertical diffusivity to the environmental factors relevant to vertical mixing such as stratification, current and wind speed. For interpretation of dispersion characteristics for practical problems, the reader is referred to the following reviews: Okubo (1971, 1974), Murthy (1976), Murthy and Okubo (1976), Kullenburg (1969) and Kullenberg et al. (1973).
Figure 4.11 Horizontal eddy diffusivity versus length scale of diffusion (longitudinal direction, after Murthy, 1976)
Figure 4.12 Horizontal eddy diffusivity versus length scale of diffusion (lateral direction, after Murthy, 1976)

The graph shows the relationship between the horizontal eddy diffusivity, $K_x$, and the length scale of diffusion, $L_x$. The data points are divided into two categories: Surface layer (up to 6 m.) and Deep water (15 m. to 50 m.). Two linear regression lines are depicted, each with a different slope:

- For the Surface layer experiments, $K_x = 1.3 \times 10^{-2} L_x^{1.3}$
- For the Deep water experiments, $K_x = 2.4 \times 10^{-3} L_x^{1.3}$

The graph uses a logarithmic scale for both axes to better visualize the data over a wide range of scales.
4.5 APPLICATION OF HYDRODYNAMIC CONSIDERATIONS TO ASSESSING POLLUTION PROBLEMS

It may perhaps help in the appreciation of the complexities of the hydrodynamic processes in lakes and reservoirs, if some practical applications are discussed, involving the direct and indirect use of lake water for the disposal of pollutants. This will be illustrated with examples for continuous and instantaneous discharges.

An important problem in the siting of thermonuclear power plants on the shores of lakes is the location of inlets for drawing water for condenser cooling and of outlets for the discharge of hot waste water. The choice is quite critical in view of the complex characteristics of the coastal currents. With periods of stagnation or very weak currents and frequent current reversals, circulation of warm water into the inlets may result in considerable power reduction. Another similar problem of practical interest is the location of sewage and other industrial waste effluent outlets and municipal water intakes. There are several engineering solutions to problems of this type. For example, one can locate the inlet for drawing water in the hypolimnion and discharge waste water in the epilimnion. While this may be a desirable solution for locations of inlets and outlets for thermonuclear power plants, it is certainly undesirable for the location of sewage effluent outlets. If not for any other reason the discharge of sewage effluent or other waste waters in the surface layer of the coastal waters is undesirable aesthetically as well as recreationally. In this case, the location of the sewage effluent outlet at greater depths and the location of water intakes outfalls located in the surface layer may be desirable. Submerged sewage outfalls located at some distance—say, 1 - 2 km from the shore are often used in oceanic coastal zones. The underlying idea is to trap the effluents below the thermocline, and allow further dispersion and transport by the hypolimnion currents and their turbulent eddies. In lakes, however, these methods may not be practical due to frequent upwelling and downwelling of coastal waters. With the discharge below the thermocline, the effluents most probably are trapped below the thermocline in the hypolimnion. The transport and dispersion processes are considerably weaker in the hypolimnion compared to the epilimnion. With the possibility of frequent upwellings and shoreward transport, very high concentrations may be brought to the surface close to the shore.

Continuous discharge methods are widely used for the disposal of pollutants in rivers and lakes. In view of its obvious practical importance, we will discuss this problem in some detail here. In order to elucidate the basic physical processes, let us consider continuous discharge of some pollutant through an outlet pipe at some distance from the shore. The initial dilution (near field dispersion) which can be achieved by a well designed diffuser is important, but this can be studied separately and, therefore, we shall not be concerned with it here. 'Far field' transport and dispersal by 'natural' processes are carried out by lake currents and their turbulent eddies and it is of practical significance in the disposal and dilution of pollutants over larger water masses.

If the rate of discharge of pollutant is \( q \) g/s and if the coastal current is steady and uniform of magnitude \( U \) cm/s, then the amount of pollutant in a cross-section of the plume which forms in the wake of the source is proportional to \( q/U \). Thus, for a continuous discharge of pollutants of given source strength, concentration in the plume will be proportional to \( U^{-1} \), a practically important result. The magnitude of current speed is an important physical parameter in the dilution of pollutants discharged continuously to the lakes. For instantaneous discharge of pollutants, the current speed does not exert a direct influence on the dispersion.

In many practical problems, one is often concerned with some sort of time mean concentration at a certain point at a certain time. Gaussian diffusion models (Okubo, 1962) have been extensively used for quantitative modelling and dilution estimates of continuous effluent plumes in steady and uniform coastal currents. Using these models one can make reasonable estimates of mean concentrations around an effluent source, given the average current speed and the persistency of current in a given direction, and with some knowledge of turbulent diffusion processes. For a continuous discharge case, a two-dimensional Gaussian model is appropriate. The maximum concentration is given by:

\[
C_m = \frac{q}{\pi S_y S_z U} \tag{4.2}
\]

where \( S_y \) and \( S_z \) are the lateral and vertical standard deviations of the plume spread. If the mean current \( U \) and the rate of discharge \( q \) are constant, then one would have to know the growth of the plume with distance from source, i.e. \( S_y(X) \) and \( S_z(X) \), to estimate downstream concentrations within the plume. In practice, one is often concerned with these concentrations at some specific distance from the point of discharge, say \( X \). For a Gaussian plume model,
the standard deviations $S_y$ and $S_z$ can be related approximately to the corresponding eddy diffusivities:

$$S_y^2 = 2K_y \frac{X}{U} \quad \text{and} \quad S_z^2 = \frac{X}{U} \quad (4.3)$$

If the initial concentration of the pollutant is $C_i$ (g/cm$^3$) and the volume rate of discharge of the pollutant is $d$ (cm$^2$/s), then the source strength of this pollutant is $q = C_id$ (g/s). Inserting this in Equation (4.2) we have:

$$C_m = \frac{C_id}{\sigma_S S_z U} \quad (4.4)$$

The 'nuisance' value of a pollutant at some distance X from the point of discharge can be described by a 'dilution' factor defined by:

$$\eta = \frac{C_m}{C_i} = \frac{d}{\sigma_S S_z U} \quad (4.5)$$

From representative values of eddy diffusivities $K_y$ and $K_z$ (for example, Figures 4.11, 4.12), one could estimate the corresponding values of $S_y$ and $S_z$. The dilution factor can be estimated knowing the magnitude of the mean current $U$ and the volume rate of discharge at the source.

The above analysis is valid if the discharge takes place in sufficiently deep water and there are no diffusion floors inhibiting vertical mixing. However, if the depth of water is limited or if a shallow diffusion floor is present at depth $h$, then vertical diffusion is limited to this depth. In such a case, it is essentially a one-dimensional problem where $\sqrt{2\pi S_z}$ in the denominator is replaced by $h$ in all equations and calculations are repeated.

4.6 SELF-PURIFICATION PROCESSES

4.6.1 Introduction

Self-purification processes in lakes and reservoirs are controlled by the hydraulic behaviour of the water mass (see section 4.2) and by a series of other important factors, namely:

- dissolved oxygen supply
- pH
- water column stability and stratification
- residence time in the littoral region
- particulate suspension
- dissolved solids, including organic matter
- temperature profiles
- atmospheric loadings
- nutrient and productivity controls
- depth and concentration gradients
- aquatic ecocommunity

In the following text, some insight is given into the most important of the above factors: oxidation, biological activity and sedimentation.

4.6.2 Oxidation

As in rivers, the major inputs of dissolved oxygen are from atmospheric reaeration, exchange mechanisms with water richer in oxygen e.g. rainfall, photosynthesis and, in some circumstances, chemical reduction of nitrate and sulphate. The major demands on the oxygen are from biological and chemical processes in the hypolimnion and sediments. The assimilative capacity of a lake and the resulting dissolved oxygen levels are normally determined as part of the overall oxygen budget. The process is similar to that used for streams but there are some important differences. Thermal stratification separates the major input (surface aeration) from the major demand (sediments). Further, in lakes both the sediment and water column demands are functions of dissolved oxygen levels in the water. Sediment demands for eutrophic waters are 0.5 to 3.0 g O$_2$/m$^2$/day and a change of 4 mg/l of dissolved oxygen doubles the demand (see Polak and Haffner, 1978).
Some measurements of sediment oxygen demand have been related to the percentage of organic matter (dry weight) for lakes. In a typical case, 1.3, 12 and 30 - 40 percent organic matter have oxygen uptake rates of 0.01, 0.1 and 0.15 - 9.18 \( \text{g O}_2 \text{ m}^{-2} \text{h}^{-1} \) at 15°C (Edberg and Hofsten, 1973). It should be noted that these rates are temperature specific and quite different uptake rates will be found at 10°C and 20°C. Further, the general applicability of these results must be tempered by sediment depth and its physical, chemical and biological quality. The final uptake will depend on whether only the surface of the sediment requires oxidation or the sediment is being disturbed so that demands in the sediment are exerted at depth. Bacterial and macroinvertebrate respiration and nitrate concentrations at the sediment-water interface may also be factors affecting oxygen uptake rates. Water oxygen demands in an urban area are 0.2 to 1.0 \( \text{g O}_2 \text{ m}^{-2} \text{d}^{-1} \) and double for an 8 mg/l rise in dissolved oxygen levels in the water (see Polak and Haffner, 1978). Both these demands are also a function of temperature, although the temperature variation in many lakes is small compared to the variation of the dissolved oxygen levels in the water.

The oxygen demand is also frequently spatially variable. Measuring changes in biological oxygen demand and chemical oxygen demand in the effluent plume of a shoreline discharge in a large lake would normally require tracing the plume for periods in excess of 12 to 14 hours (Polak and Palmer, 1977).

Atmospheric reaeration is highly variable and difficult to measure. Normally all the other inputs and demands are measured and the reaeration determined by difference. In most instances reaeration ranges from 1 to 9 \( \text{gO}_2/\text{m}^2/\text{day} \). Some typical oxygen budgets are shown in Table 4.2.

### Table 4.2 Typical oxygen budgets

<table>
<thead>
<tr>
<th>Oxygen inputs and demands</th>
<th>Hamilton Harbour (Summer)</th>
<th>Thames River Estuary (Owens, 1964)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>INPUTS (percentage)</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Atmospheric reaeration</td>
<td>68</td>
<td>69</td>
</tr>
<tr>
<td>Lake/ocean exchange</td>
<td>19</td>
<td>8</td>
</tr>
<tr>
<td>Tributary</td>
<td>--</td>
<td>9</td>
</tr>
<tr>
<td>Photosynthesis</td>
<td>13</td>
<td>9</td>
</tr>
<tr>
<td>Rainfall</td>
<td>--</td>
<td>0.5</td>
</tr>
<tr>
<td>Reduction NO₃ &amp; SO₄</td>
<td>--</td>
<td>4.0</td>
</tr>
<tr>
<td><strong>DEMANDS (percentage)</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Epilimnion</td>
<td>30</td>
<td>not applicable</td>
</tr>
<tr>
<td>Metalimnion</td>
<td>27</td>
<td>&quot;</td>
</tr>
<tr>
<td>Hypolimnion</td>
<td>26</td>
<td>&quot;</td>
</tr>
<tr>
<td>Sediment</td>
<td>13</td>
<td>&quot;</td>
</tr>
<tr>
<td>Change of the stock</td>
<td>4</td>
<td>&quot;</td>
</tr>
</tbody>
</table>

The maintenance of reasonable dissolved oxygen levels allows the conversion of potentially biologically toxic chemicals like \( \text{H}_2\text{S} \) and ammonia into less harmful components. These oxidations are normally very rapid and usually are a function of the dissolved oxygen stock available. Depending on the length of thermal stratification, reaeration of the hypolimnion may be impossible for months. Full oxygen depletion and significant concentrations of \( \text{H}_2\text{S} \) will persist for long periods as can be seen in the example of Lake Kariba (Figure 4.6). In water free of oxygen above the sediments and within the sediments, remobilisation especially of iron and manganese occurs. The presence of either of these metals causes great difficulties for drinking water supply. They form deposits in the pipes and removal requires expensive treatment.

Discharges of free chlorine are normally reduced rapidly in a natural environment provided ammonia is not present, in which case toxic chloramines are formed. Very low oxygen levels allow reducing conditions to establish which can release nutrients from the
sediments particularly phosphorus thereby enriching the water column above at the time of overturn. A water column with alternating oxic and anoxic conditions at the sediment interface can act as a nutrient sink for nitrogen through nitrification/denitrification (Keeney, 1973).

4.6.3 Biological activity

If slightly polluted water courses are impounded, the biological activity becomes much more intensive both in time and space than it was in flowing water, i.e. the degradation effect on the constituents of the waste water is increased. This effect is utilized in drinking water reservoirs, which, if properly dimensioned have such biological activity that a considerable improvement in water quality is achieved (Lack and Collingwood, 1975).

The material budget of storage reservoirs and lakes is governed by the phytoplankton. The production of algae is a function of nutrient availability, light and efficiency of its utilization, grazing intensity of the water column and in some instances, the presence of toxins or parasites. In many instances, algal production can be related to total phosphorus loading (Vollenweider, 1968, Vollenweider and Dillon, 1974) but predictions of the effects of phosphorus on algal production have been questioned (Thomann, 1977) because of the interactions of other variables. The effects of these other variables have been expressed in production models (Bannister, 1974 and Lehman et al., 1975) which may be better ways to produce predictive statements on algal production. As more variables are considered the prediction becomes more realistic. A model especially developed for shallow lakes was described by Oskam, (1973).

As a result of eutrophication, production rates are often accelerated, resulting in high standing crops of algae and locally high concentrations of dissolved oxygen. However, the eventual decomposition of these populations results in high oxygen demands being exerted on the water column. Should these demands exceed rates of oxygen supply, anoxic conditions will arise.

Extensive zones of macrophytes have some effect on the concentration of nutrients and degradation of pollutants carried by the incoming water. However, their presence also involves problems of silting and the accumulation of persistent pollutants, particularly heavy metals, in the sediments.

Bacteria are an important component of the bioactivity in lakes and reservoirs (Golterman, 1975). Consequently, productivity estimates for bacteria should be made. Unfortunately little information is available on the nutrient requirements for bacteria in general other than some efforts to correlate bacterial populations with some nutrients. Heterotrophs, nitrosomonads and nitrifying bacterial populations are important in the assimilation of nutrients. The presence of pathogenic bacteria limits water use for both drinking and body contact recreation. Estimates of bacterial populations are prone to at least three major influences. The time of sampling is critical as Bellair et al. (1977) have shown that sunlight can produce a two-fold variation in population numbers within a day.

Water temperature is a further important influence. Geldreich (1968) has shown that Salmonella typhimurium, Escherichia coli and Aerobacter aerogenes were reduced by 90% after 1.3, 1.9 and 3.8 days respectively when stored in storm water at 20°C, while faecal streptococci were only reduced by 83% after 14 days. At 10°C the 90% reduction in numbers took 7.6, 9.3 and 5.8 days respectively. Faecal streptococci were reduced by 48% after 14 days at this temperature.

Distance and time of travel from the discharge is another influence to be taken into account (Zanoni et al., 1978). The decay of the bacteria with distance is normally computed by considering the time of travel and water temperature. Receiving bodies of water vary in their characteristics which affect the decay rates and the extent must be assessed with actual field measurements. To permit an assessment of this variation, bacterial surveys are normally required over a period of time (approximately 5 days) with at least two samples per day. Data from these surveys are useful for developing decay rates for the bacteria. Generally, bacterial decay is a power function of distance from the source. Typically, reductions of bacterial levels by a factor of two orders of magnitude occur in a kilometre along the shoreline of large lakes (Cherry et al., 1974).

Perhaps the most important roles of bacteria are their part in the assimilative capacity of the body of water. They break down large organic molecules, help stabilize organic content in sediments, and even break down harmful toxins. This assimilative capacity generates high oxygen demands. Stabilization of sediments can result in an oxygen
uptake rate in the range of 0.05 to 9.18 g O₂ m⁻² d⁻¹ at 20°C and the water column demands are frequently greater.

There is evidence that bottom fauna assimilates insecticides, polychlorinated biphenyls, heavy metals and radioactivity. This fauna is the major source of these substances in fish, either directly or through predation. While the bottom fauna is probably not a significant mechanism for the removal of these substances from water compared to other mechanisms, it is a good indicator of the degree of contamination.

4.6.4 Sedimentation

Pollutants discharged to nearshore waters by industries and sewage treatment plants are either transported in dissolved, complexed or suspended form to offshore waters or sedimented by various competing mechanisms. Numerous studies have been performed on the mechanisms of trace metal transport and sedimentation in rivers, lakes and estuaries. Important mechanisms of sedimentation (Gibbs, 1973; Stumm and Morgan, 1970) include incorporation in inert crystalline structures such as various silicate minerals; precipitation and coprecipitation as oxides, hydroxides, carbonates, sulphides etc; absorption (physical and chemical) on minerals such as clays; or biological incorporation in sedimentation.

The chemical nature of the sediment-water interface plays a profound role in the sedimentation of pollutants and their possible re-solution. It has been known for many years (Mortimer, 1941) that anaerobic conditions in the overlying water allow reduction and mobilization of absorbed or co-precipitated phosphate and silica. Other factors governing the mobilization of trace metals include:

- Increased salinity, particularly in estuaries, may lead to competitive adsorption of seawater cations (Forstner, 1976).
- Decreased pH increases the solubility of carbonates and hydroxides.
- Increased use of synthetic complexing agents such as nitrilotriacetic acid (NTA), which allows the formation of heavy metal chelates that remain in solution.
- Microbial activity can affect the physical and redox properties of sediments bringing about reducing conditions. Bacteria are also involved in the formation of soluble organo-metallic compounds (e.g., with mercury).

An example may be drawn from the Hamilton Harbour study (Ontario, Ministry of the Environment, 1974) to illustrate the importance of these sedimentation mechanisms on the control of heavy metal concentration in water. If cultural and natural inputs to Hamilton Harbour, but not lake exchange, are considered the average residence time for water in the harbour is 1.25 years. Using this figure expected metal concentrations were computed for iron, chromium and zinc from industrial data and compared with the observed concentrations. The fraction remaining in solution is generally 5 per cent or less. Although some of the metal may have been removed by lake exchange, the majority is undoubtedly in the sediments. The greatest concentration of heavy metals is found in deep water sediments or adjacent to the discharges. These produce enrichment of surface layers of the sediment compared to deeply buried layers. Average enrichment factors (top/bottom concentration ratios in cores) of up to six have been found in Hamilton Harbour. In Lake Erie, Walters et al. (1974) found concentration factors of 2 to 50 for different metals.

Contamination of deep water sediments is indicative of polluting discharges. However, determining the pattern of contamination and relating it to the discharge is difficult. It is suggested that if contamination is indicated in the sediments or biological species, discharges be sampled and analyzed. If discharges cannot be identified, atmospheric fallout (GESAMP, 1976) should be examined as well as natural sources like groundwater.

4.7 MAJOR KINDS OF BIOTIC CHANGE CAUSED BY SELECTED POLLUTANTS

In Chapter 2 the sources of the major eutrophicating elements nitrogen and phosphorus were identified and the problems of eutrophication have been briefly described earlier in this chapter. The problem will now be discussed in more detail with regard to consideration of eutrophication as a pollution process. The opposite processes of inhibition and toxicity are briefly surveyed.

4.7.1 Substances causing eutrophication

Eutrophication is the excessive fertilization of a water body which results in the abundant growth of aquatic plants such as algae or macrophytes. The increase in nutrients in a water body can occur naturally in the normal evolution of a lake but the process is accelerated by
human activities. The subject has been studied in great detail and the reader is referred to specialized texts such as Vollenweider 1968, Vollenweider and Dillon 1974, Golterman 1975, and Vollenweider 1980.

In most temperate lakes in the world, it is the phosphorus compounds which cause eutrophication, nitrogen being present in excess quantities. In tropical lakes, nitrates have been found to be responsible for causing eutrophication.

The main sources of nitrogen and phosphorus causing eutrophication are:

1. Community wastewater (excreta and detergents)
2. Drainage from agricultural land, forests and wasteland
3. Input from animal excrement
4. Erosion products
5. Precipitation; wet and dry
6. Storm water drains
7. Internal recycling within the waterbody.

The consequences of eutrophication can be separated into chemical and biological. Briefly, the chemical consequences of eutrophication are:

1. pH shifts to alkaline as CO₂ is utilized
2. The oxygen resources are changed. There may be excessive production in the trophogenic layer and complete utilization in the tropholytic layer resulting in deoxygenation.
3. This brings about reducing conditions which favour incomplete mineralization of organic substances, reduction of nitrate to nitrite and ammonia, conversion of sulphates to hydrogen sulphide, release of iron and manganese from the sediments to the free water, formation of iron sulphide, release of phosphate from the sediment to the free water.

This last mentioned consequence is also termed 'internal loading' and is an important factor in the continued eutrophication of water bodies following reduction in the external sources of nutrients.

The biological effects of eutrophication which are apparent to the layman are the excessive growths of algae and aquatic plants. Filamentous green algae such as Cladophora may be abundant in the littoral zone and may blanket any other kind of plant. The spread of floating-leaved plants such as Potamogeton may also be very obvious.

Oligotrophic lakes are characterised by a scarcity of plankton. Diatoms and Chrysophyceae will occur but their population density is low. Eutrophication manifests itself first in a rise in cell numbers particularly of diatoms such as Tabellaria and Asterionella and most of the Chrysophyceae (eg, Dinobryon) will disappear from the plankton. Further eutrophication results in a greater increase in the numbers of diatoms, the appearance of green algae and some blue-green algae of the genus Oscillatoria may produce conspicuous water blooms. Finally, blue-green algae such as Anabaena, Aphanizomenon and Microcystis will dominate for most of the late summer and autumn.

In Lake Constance the following sequence of events took place. Until 1950 phosphorus was not detectable in the lake water. Algal density was low and the development of zoo-plankton was equally poor. Beginning with the large scale use of fertilizer and detergent, the phosphate concentration and, simultaneously, the phytoplankton and zooplankton production increased. Fish also grew faster and reached larger sizes. The spectrum of algal species shifted from the originally predominating Chrysophyceae and pennate diatoms (Synedra acus and Tabellaria) towards centric diatoms (Cyclotella), Cryptophyceae (Rhodomonas) and Chlorophyceae (green algae). At the same time the seasonal periodicity of the plankton changed. In the initial phase of eutrophication (1953-1962), phytoplankton maxima occurred in April-May and September-October. Since then, with increased phosphate inflow, a third maximum is observed in July-August (Lehn, 1975).

The effects of eutrophication on zooplankton generally manifest themselves in an increase in the numbers of herbivorous species in response to the richer and more varied food supply. The size of the algae plays an important role because small algae have a positive effect on the development of small herbivorous members of the zooplankton such as Rotatoria and immature Copepoda. Only the larger grazers such as Daphnia are capable of feeding on the large algae. An interesting exception to this general statement is provided by the small cladoceran Chydorus sphaericus which is not normally found in the zooplankton. Intensive growths of the large, filamentous, blue-green alga Oscillatoria rubescens leads to the formation of large algal clumps in which Chydorus finds shelter and food thus allowing it to appear in large numbers in the pelagic zone (Brooks, 1969).
Eutrophi cation also has an influence on the fish fauna. Increased nutrient supply brings about increased algae and zooplankton. These are an important food source for small fish and the fry of most fish. Generally, eutrophication brings about an increase in fish production. Reductions in the oxygen concentration at night can cause fish mortalities in extreme cases and reduction on oxygen concentration in the hypolimnion may restrict the movements of salmonid fish and favour the less valuable cyprinid species. The high pH values created by mass growths of blue-green algae may also cause distress or mortality.

4.7.2 Inhibiting or toxic substances

Although toxic effects in water can obviously be extremely serious, they are difficult to describe in a general way. There is a multitude of substances which are either directly toxic or may become acutely toxic under certain conditions.

Where toxic substances such as, for example, cyanides, are directly discharged into surface waters their harmful effects on aquatic organisms can be easily recognized from the resulting fish mortality. It is far more difficult to recognize the effects of the introduction of toxic substances which do not lead to directly visible catastrophes. Within the overall ecological effects of toxic substances, various individual effects which may be synergistic or antagonistic, are superimposed on each other. The most important individual effects are listed below:

1. With rising water temperature, the metabolic activity of the organisms intensifies, thus increasing their sensitivity to acutely toxic substances.
2. The chemical characteristics of the receiving water body influences the toxicity of polluting substances. For example, the toxicity of zinc ions decreases with increasing water hardness.
3. Low concentrations of toxic substances which produce no acute effect may lead to a certain degree of adaptation so that higher concentrations can be endured.
4. The reactions of organisms to toxic substances vary in the course of their life cycles. Generally, the younger stages (eggs, larvae, fry) are more sensitive than fully grown animals. There are also gradations within the same group of organisms: salmonid fish are generally much more sensitive than cyprinid fish. Even with fully grown animals of the same species, the physiological condition of the individual is of considerable importance in determining the actual effective toxicity threshold of a substance.
5. Some substances are highly toxic but are very quickly hydrolyzed or degraded in water so that they are effective for only a very short time period (eg, parathion).
6. Various toxic substances, in particular the heavy metals, have synergistic effects. For example, if copper and zinc combine, they are much more toxic than the sum of their individual toxicities.
7. Of particular importance are toxic substances that are not subject to biological degradation but are stored in the organisms and may finally reach the human body on their passage through the food chain, via algae, lower animal species and fish. In this respect, heavy metals, chlorinated hydrocarbons, polychlorinated aromatics (PCA) and polychlorinated biphenyls (PCB) are particularly significant. Due to their accumulation along the food chain, their use in pest control often involves a serious threat to the functioning of the ecosystem and, finally, to the humans who consume the food.

A visible effect of toxic substances on aquatic organisms is the impoverishment of the aquatic biota resulting from the introduction of biologically degradable organic substances. However, as the absence of organisms may also have other causes for example, wash-off by a preceding flood, the impoverishment in organism stock is no more than a hint that toxic effects may possibly exist. Reliable proof requires special toxicological tests where the effects of the substances in question, or of the polluted water, are studied in comparison to non-polluted samples under clearly defined conditions.

In addition to the loss of a part of the aquatic life, usually fish food, the inhibition of bacterial activity must be considered as a serious consequence in surface waters. Reduction in bacterial activity means reduction in self-purification capacity (Knopp, 1968). Detailed information may be found in the WHO Reference Book on Health Hazards from the Environment (WHO 1972).

REFERENCES


