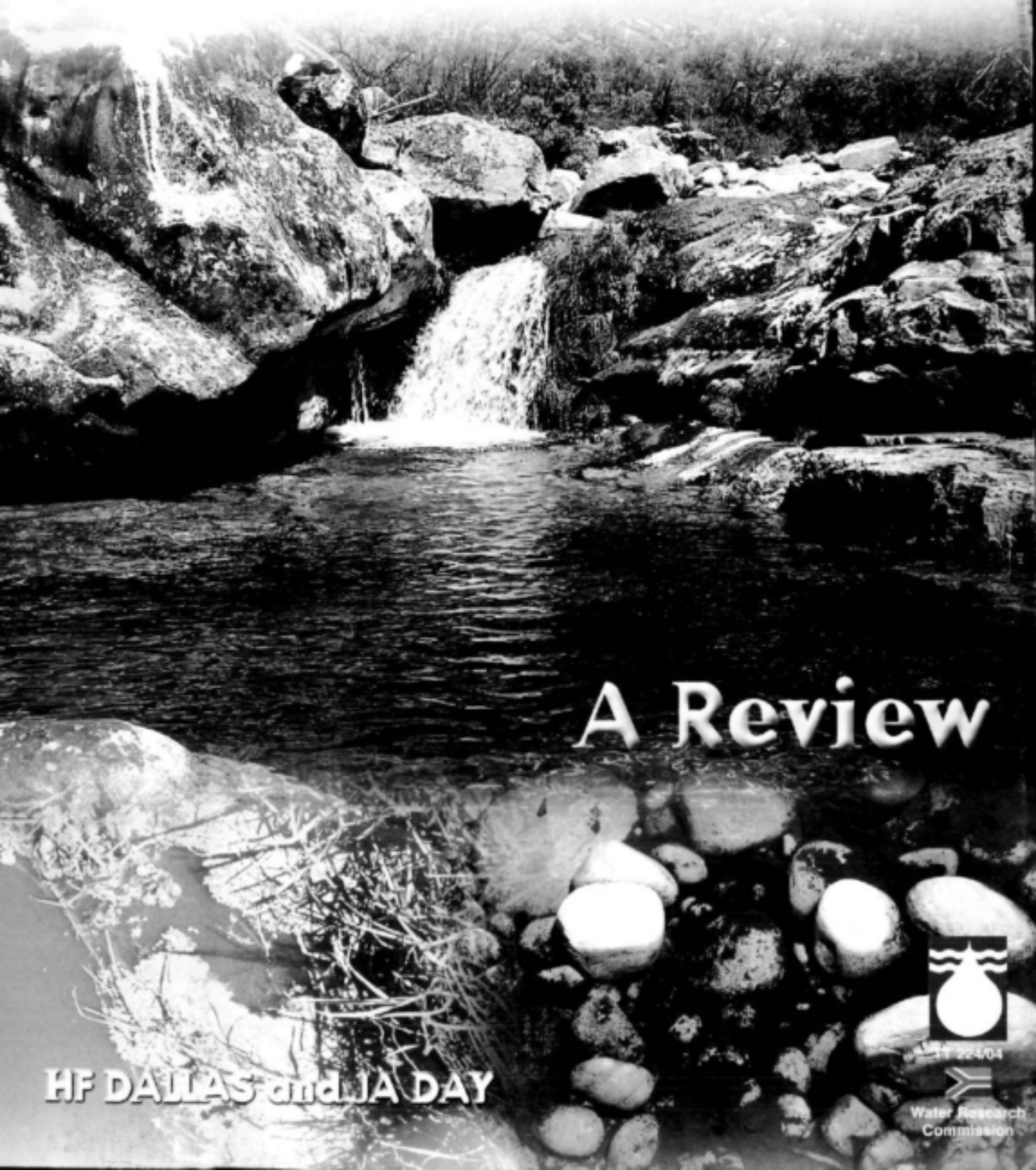
The Effect of Water Quality Variables on Aquatic Ecosystems:



A Review

HF DALLAS and JA DAY

TT 224/04

Water Research Commission

THE EFFECT OF WATER QUALITY VARIABLES ON AQUATIC ECOSYSTEMS:

A REVIEW

HF DALLAS and JA DAY

Report to the
Water Research Commission
by
Freshwater Research Unit
University of Cape Town
Rondebosch 7700

WRC Report No. TT 224/04
FEBRUARY 2004

Obtainable from:

Water Research Commission
Private Bag X03
Gezina
0031

The publication of this report emanates from a project entitled: *The updating of TT 61/93: The effect of water quality variables on aquatic ecosystems: a review (WRC Consulting No.K8/455)*

ISBN No. 1-77005-131-7
Printed in the Republic of South Africa

TABLE OF CONTENTS

CHAPTER 1: INTRODUCTION

THE NEED FOR WATER MANAGEMENT

The comparatively high temperatures and seasonal rainfall over much of South Africa mean that fresh water is a scarce resource. Permanent bodies of standing fresh water are virtually non-existent, leaving rivers as the only exploitable sources of fresh surface water. Although maps show numerous rivers in South Africa, many of these rivers are small, or flow only during the wet season, or both. Furthermore, the increase in human populations has resulted in intense pressure on South Africa's rivers, both as sources of water and in terms of pollution, and yet water is of no use to humans or for the maintenance of natural riverine ecosystems if its quality is poor. Thus water is a limited resource and both availability and quality need to be managed carefully.

Aquatic ecosystems are particularly susceptible to pollution. Rivers are confined, uni-directional systems that act as "drains" for the landscape, while lakes and wetlands are usually "sinks", accumulating materials brought in by wind, water and humans from their surroundings. Activities anywhere in its catchment are reflected in a river and its associated ecosystems and alterations or perturbations, even in the upper reaches, may have an effect down its entire length. Wetlands, being places where sediments and water tend to accumulate, are even more exposed to build-up of pollutants both in their waters and in their sediments.

The longitudinal and temporal effects of pollutants in a river depend on the extent to which its biota can degrade and so remove the particular pollutants entering it. At the same time, the extent to which the biota can purify water in a river depends on the quantity and type of pollutants entering it. No organisms can break down chemical elements like heavy metals, for instance (although they can break down and so remove most organic substances); nor can they function adequately in the presence of toxins. The effectiveness of the purification process is also influenced by the degree to which a pollutant is diluted by rainfall or river flow. Thus concentrated pollutants may inhibit or even destroy the ability of a river or wetland to cleanse its own waters.

The human population of South Africa relies almost entirely on rivers and aquifers for water. Thus until recently all of the policies of the Department of Water Affairs and Forestry (DWAF) regarding management of water quality referred almost exclusively to rivers. The South African Water Act 54 of 1956 relied on designation of effluent standards for controlling pollution in rivers but in the last decade or more, emphasis has shifted to the adoption of receiving water quality objectives (RWQO), allocating waste loads on a river-by-river and catchment-by-catchment basis (Van der Merwe & Grobler 1990). This approach has become enshrined in law with the ratification of the new National Water Act 38 of 1998. The Water Act requires that for each significant water resource (river, wetland or aquifer), a 'reserve' of water is kept aside. This 'reserve' is 'the quantity and quality of water required a) to satisfy basic human needs ... and b) to protect aquatic ecosystems in order to secure ecologically sustainable development and use of the relevant water source. Thus water quality management objectives are still based on specific user requirements, evaluated and measured against present and potential catchment conditions. The significant point is that the aquatic ecosystems themselves are considered to be *the* primary 'users' of the water they contain.

One of the aspects of water quality management in South Africa that might be of particular interest to the reader is the development of water quality guidelines for the natural environment. Interim guidelines were developed some years ago (South African National Water Quality Guidelines 1996) and can be consulted on the DWAF website at www.dwaf.org.za.

The aim of the present review is to synthesize documented research on the effects of various aspects of water quality on aquatic ecosystems. The scarcity of published literature in South Africa has meant that much information in this review has been obtained from overseas publications. Indeed, research on the effects of pollutants on ecosystems, rather than on species, is limited world-wide. It should be stressed that overseas conditions are often not the same as those pertaining in South Africa so that *extrapolation from overseas data to the local situation is not an adequate substitute for local research.*

FORMAT OF THIS REVIEW

Chapters 2 and 3 in this review provide a general introduction to the issue of water quality in relation to aquatic ecosystems. The first briefly describes aquatic ecosystems and their biotas, and how they react to changes in water quality. The second examines the ways in which these responses can be quantified. The bulk of the review (Chapters 4 - 12) synthesises what is known about the effects on aquatic ecosystems of individual chemical constituents and physical attributes of water. The last eight chapters examine the effects of different types of whole effluents, rather than of individual constituents, on aquatic ecosystems.

CHAPTER 2: AQUATIC ECOSYSTEMS AND WATER QUALITY

SUMMARY

In this review, water quality is taken to be the combined effect of the physical attributes and chemical constituents of a sample of water for a particular user. Functional aquatic ecosystems usually support a variety of organisms, such as primary producers, primary consumers and secondary consumers, within different trophic levels. Rivers are longitudinal systems driven by largely by the flow of water and are divided into zones, which are distinct with respect to their physical, chemical and biological characteristics. Wetlands are depositing systems that accumulate sediment and include a wide variety of aquatic ecosystems from riverine floodplains to high-altitude rainpools and from tree-covered swamps to saline lakes. Regional differences in rivers and wetlands arise as a result of difference in climate (and thus temperature, mean annual precipitation, mean annual evaporation, etc.), geomorphology (gradient, erosion), geology and biota. These differences need to be considered when establishing guidelines for the protection of aquatic ecosystems. Within each region or zone, community composition is determined by water quality, the type of habitat (biotope) available, the degree of water movement, temporal variations in the availability of water, and the historical distribution of species. Water quality variables potentially affecting aquatic ecosystems may be physical (turbidity, suspensoids, temperature) or chemical (non-toxic: pH, TDS, salinity, conductivity, individual ions, nutrients, organic enrichment and dissolved oxygen; and toxic: biocides and trace metals). Each variable has an effect, either beneficial or detrimental, on aquatic organisms; and the overall effect when more than one variable is involved is dependent on whether they act synergistically or antagonistically. The effect of each variable on individual organisms is also influenced by the tolerance limits of the organism. In addition to individual variables, aquatic ecosystems are often the ultimate receivers of whole effluents, which consist of a combination of water quality variables.

2.1 INTRODUCTION

This chapter outlines the ways in which water quality is affected by natural features such as geology and climate, the effects of differences in natural water quality on aquatic biotas, and some aspects of the biology and ecology of aquatic organisms. The reader is referred to Davies & Day (1998) for a more detailed coverage of these general limnological features.

Some definitions

In this review, *water quality* is taken to be the combined effect on a 'user' of the physical attributes and chemical constituents of a sample of water. The idea of water "quality" is a human construct, implying value or usefulness, and indeed the quality of any sample of water depends on the point of view of the user. Water quality from the point of view of aquatic ecosystems as 'users' is discussed in section 2.7 below.

A *water quality variable* is any of those attributes or constituents that vary in magnitude and whose variations alter water "quality".

Biota is a collective term for all living organisms, including plants, animals and micro-organisms, living in a particular ecosystem.

A *biological community* is a well-defined assemblage of interacting organisms, such as the community of organisms living in the stony bed of a river or the species of fish living in an estuary.

An *ecosystem* consists of interacting organisms plus their environment. Ecosystems can be as simple as human skin, with its interacting microflora; or a jar of water taken from a pond; or the pond itself; or they can be as complex as a whole river with its riparian vegetation and the animals dependent on that vegetation.

With regard to rivers and wetlands, the *natural environment* can best be defined as aquatic ecosystems and those ecosystems dependent on them. It includes not only the submerged ecosystems of rivers and wetlands but also the associated riparian belts and floodplains, and the wildlife that depend on rivers for drinking water and on riparian vegetation for shelter. It should be noted that although we use the term '*natural* environment', few ecosystems in South Africa are truly unaffected by human activity.

2.2 THE STRUCTURE OF AQUATIC COMMUNITIES

Any biological community is made up of a number of interacting species that can, for convenience, be grouped into *trophic levels* (*trophic* meaning "feeding"). The various species of *plants* or *primary producers* form the basic trophic level in most well developed natural communities. These in turn are fed upon by the *primary consumers* or *herbivores* (a second trophic level), which in turn are fed upon by the *carnivores*. Carnivores feeding on herbivores are said to be *secondary consumers*, those feeding on secondary consumers are said to be *tertiary consumers*, and so on.

Dead and decaying organic matter, collectively known as *detritus*, is broken down by *decomposers*, which include scavenging and *detritivorous* (detritus-eating) animals, and also *decomposer microbes* such as bacteria and fungi. A functional aquatic ecosystem usually supports a number of different species of each of these kinds of organism. The ultimate effect of the interactions between these organisms is the breakdown of dead organic matter (which may include fallen leaves and branches, dead bodies, decaying phytoplankton or the organic components of sewage). In the process, nutrients are *mineralized* and released back into the ecosystem in simple inorganic form. The combination of decomposition and mineralization results in purification of the water. Thus healthy, functional ecosystems provide a significant service for human beings by cleansing the water in the rivers in which they occur. An important corollary is that when ecosystems are subject to excessive levels of pollutants, the decomposers themselves are detrimentally affected and can no longer provide this service.

2.2.1 Rivers and wetlands as ecosystems

In order to manage any ecosystem successfully, a good knowledge of its basic functioning is essential. Further, it is possible to ascertain the effects of human interference (e.g. pollution, impoundments, etc.) on an aquatic ecosystem only if we have information on its natural characteristics and on the ways in which its biota responds to change. Below we describe the most fundamental features of rivers and wetlands.

2.2.1.1 Rivers

The following is a description of a generalized river ecosystem (see also Table 2.1). The driving physical factor is the flow of water. Current speed is highest in the upper reaches and declines steadily downstream as the river descends from the highlands and the gradient reduces. Most rivers can thus be divided into three distinct zones: a fast-flowing erosive headwater zone; a slower-flowing, partly-erosive middle zone; and a slow-flowing lower reach or mature river where materials eroded from the upper reaches are deposited.

The Headwater Zone or mountain stream

A typical headwater stream is characterized by clear, fast-flowing, well oxygenated, nutrient-poor water, a boulder-strewn bed and abundant, canopy-like riparian vegetation that reduces light penetration and thus limits the number of plants that can grow in the water. Food for the aquatic animals (fish and invertebrates) consists largely of detritus: leaves, fruit, and so on that have fallen from the trees overhead. This material is gradually broken down from coarse to fine particles by detritivores and decomposer micro-organisms. Headwater reaches are dominated by organisms like the nymphs or larvae of mayflies (Ephemeroptera), stoneflies (Plecoptera) and caddisflies (Trichoptera), as well as both young and adult stages of beetles (Coleoptera) and bugs (Hemiptera) adapted to living in fast-flowing water (Davies & Day 1998). The most numerous invertebrates are the shredders, which feed by breaking up or shredding leaves and other coarse organic particles, although predators, and collectors of particles, will also be present. Food is usually scarce, resulting in slow growth rates and ultimately in small adult invertebrates. Generally the species found in mountain streams are able to survive only within a very narrow range of environmental conditions, so that alterations to water quality in this region may have profound effects not only here but in the river ecosystem as a whole.

The Middle Zone

In the middle reaches, the stream widens, the current speed and oxygen content decrease, and temperature and turbidity may increase. Because the river is wider the riparian belt of trees do not form a closed canopy overhead but allow sunlight to reach the water, resulting in an increase in photosynthesis and hence in biomass of submerged plants. The food for the aquatic animals thus consists mostly of algae and aquatic plants generated within the stream itself. The invertebrate community is dominated by collectors, which gather fine particles of food by erecting nets, sieves or strings of saliva that filter the water, and grazers, which feed on the algal layer covering the rocks.

The Lower Zone: the mature river

In the lower reaches, the river continues to widen and flow rates to decrease, resulting in the settling out of fine organic particles, which cover stones and form a relatively uniform sandy or silty layer on the river bed. The water is less oxygenated, while mineral and nutrient loads increase, leading to increased plant growth. Increased sunlight and slower flow rates encourage the growth of phytoplankton (minute suspended algae) and zooplankton (minute suspended animals). Collectors, including deposit feeders (which consume the organic material deposited on the river bed), also become abundant.

It is necessary to remember that this is a generalized account of a river ecosystem and that there are always exceptions to the basic pattern. Local exceptions include rivers that rise in coastal hills and plunge into the sea along the southern and south-western Cape; mature rivers that flow over a second mountain range near the coast and are "rejuvenated" (e.g. the Augrabies Falls on the Orange River); the naturally high silt load of the Orange River; and the acid waters of rivers draining the ancient, weathered, nutrient-poor south-western Cape area (Dallas *et al.* 1998, Day *et al.* 1998). Because rivers are not all the same, it is important to be aware of the type of river and its basic characteristics when assessing the water quality required by its biota.

Table 2.1 Generalized characteristics of each riverine zone. The extent to which specific characteristics change down the course of a river is dependent on land-use practices and pollution sources along its course, and changes are relative with respect to each zone.

Characteristic	Headwater Zone	Middle Zone	Lower Zone
Physical			
Slope	Steep	Gradual	Gradual
Velocity	Fast, erosive	Slower	Slow, depositing
Substratum	Mainly boulders	Mixed boulders and sand	Sand and mud
Temperature	Normally low	Slightly warmer	Warm
Turbidity	Clear, low	Intermediate	High
Light reaching stream	Low	Intermediate	High
Riparian vegetation	Canopy-like	Concentrated on banks	Only on banks; often exotic species
Chemical			
Dissolved oxygen	High	Intermediate	Low
Nutrients	Low	Intermediate	High
Conductivity	Generally increases down the river		
pH	Generally increases down the river		
Biological			
Energy input	CPOM[1]	CPOM & FPOM[1]	FPOM
Plankton[2]	Mostly absent	May be present	Often present
Dominant invertebrates	Shredders, collectors	Collectors, grazers	Collectors

1: CPOM = Course Particulate Organic Matter; FPOM = Fine Particulate Organic Matter

2: *Plankton consists of the small plants (phytoplankton) and animals (zooplankton) that live suspended in the water column*

2.2.1.2 Wetlands

If rivers are eroding systems that transport particulate material, then wetlands are depositing systems that accumulate suspended material. Thus the bed of a wetland is almost invariably composed of fine muds or sands rather than rocks and cobbles. The term 'wetland' covers a wide variety of aquatic ecosystems from riverine floodplains to high-altitude rainpools and from tree-covered swamps to saline lakes. The determinants of wetland conditions are hydrology (the distribution of water over time and space) and geomorphology (the shape of the land): the presence or absence of water from season to season, the level of the water when it is present, and the presence of depressions or basins where water can accumulate. Because the bottom usually consists of soft sediments, wetlands exhibit fewer different types of biotopes than rivers do. The major aquatic biotopes for invertebrates are the open water, inhabited by suspended planktonic forms and fish; the shallow edges, inhiabited by benthic (bottom-dwelling) forms; and marginal, rooted and floating plants inhabited by yet other assemblages of invertebrates and fish. Major differences in physical and chemical conditions are seen vertically (i.e. through the depth of the water column) rather than longitudinally, as they are in rivers.

2.3 THE EFFECTS OF WATER QUALITY ON AQUATIC ECOSYSTEMS

2.3.1 Regional differences in water quality

Water quality differs from continent to continent, and even from region to region, as a result of differences in climate, geomorphology, geology and soils, and biotic composition.

Climate affects water quality in a number of ways. For instance, *temperature* is not only an attribute of water quality *per se* but it also determines the rate and extent of various chemical interactions (see Table 2.2). *Mean annual precipitation*, and *seasonal variations* in precipitation, determine the amount of water flowing in rivers at different times and therefore dictate the degree of dilution of natural and anthropogenic constituents in the water. *Mean annual evaporation*, as well as *seasonal variations* in evaporation, have the converse effect, concentrating the constituents in water and, if sufficiently intense, altering the proportions of major ions and ultimately resulting in precipitation or crystallization of salts.

The ***geomorphology*** of the landscape determines, amongst other things, the *gradient* of rivers (and hence the amount of turbulence and thus the amount of oxygen and other gases dissolved in water) and the degree of *erosion* (and hence the turbidity and the quantity of suspended material in the water).

Because ***geological*** formations consist of rocks of different chemical composition, they and the soils derived from them contribute different quantities and different proportions of ions (including nutrients) to the waters flowing over or percolating through them.

Various components of the ***biota***, particularly phytoplankton and microbes, can also profoundly affect water quality. The combined effects of photosynthesis and decomposition, for instance, can determine both the pH and the amount of oxygen present in water. Catchment vegetation, too, may produce organic compounds that, when leached into water, affect pH and inhibit microbial activity.

2.3.2 Regional differences in water quality in South Africa

South Africa is diverse in climate, geomorphology, geology and soils, and aquatic biotas, and so the different regions exhibit quite considerable differences in water quality even when unaffected by human activity. The descriptions below refer to what appears to have been the "natural" situation While few rivers have been entirely unaffected by industrialization and other human activities, most retain some of the features discussed in the following section.

Because of differences in climate and geomorphology, the southern and eastern parts of the land are more mesic (i.e. with a more equitable climate) than is the rest of the country. Thus their rivers tend to be perennial and to carry fairly pure water low in total dissolved solids (TDS). Except for the Orange River and some of its tributaries, rivers of the arid interior, and of the "arid corridor" of the eastern Cape, may be seasonal or ephemeral. During hot, dry periods their waters may undergo evaporative concentration and TDS values may increase by an order of magnitude or more. South Africa is geologically diverse (see, for instance, Tankard *et al.* 1982 for details). The great sedimentary Karoo basin is surrounded to the north largely by a complex of igneous formations and to the south and east largely by the sandstones and shales of the Cape and Malmesbury Supergroups. The youngest of the Karoo Series is the basalt cap (also igneous) of the Drakensberg.

Igneous rocks, such as those of the Transvaal and the Drakensberg, usually contain sufficient calcium and magnesium that water flowing over or through them picks up measurable quantities of these elements, and of nutrients such as phosphates, nitrates and silicates. In all, water affected by igneous rocks is usually dominated by calcium and/or magnesium cations and bicarbonate anions and has a pH higher than 7. Conductivity is usually low (<50 mS m^{-1} often <20 mS m^{-1}). Such waters are said to be "rock dominated" (Gibbs 1970).

At the other extreme, sedimentary rocks, such as those of the Cape Supergroup, may have been derived from particles that were already strongly weathered when they were consolidated into rock. Very little soluble material is present and so even less can be leached out. Waters flowing over such rocks are usually very low in conductivity and in nutrients. The bulk of dissolved material is derived from rain, snow and other forms of precipitation, in which the major ions are sodium and chloride. Such waters are said to be "precipitation dominated" and are very soft, pure and unbuffered. The pH is determined by the concentrations of the components of the carbonate/bicarbonate buffering system (see Chapter 6) and of any organic acids that may be present, and in different systems normally varies from very acid (pH <4) to slightly alkaline (pH <8).

In summary, the major chemical properties of any natural body of water are determined by the combined effects of geology and climate (see Day & King 1995 with regard to South African water chemistry)..

2.4 THE EFFECTS OF SOME PHYSICAL ATTRIBUTES AND CHEMICAL CONSTITUENTS ON AQUATIC ORGANISMS

The actual species of organisms that comprise any aquatic biological community are largely determined by:

- water quality

- the types of biotope available (e.g. stones-in-current, marginal vegetation)
- the degree of movement of the water (e.g. lakes *vs* rivers) and temporal changes (e.g. periods of spate and drought)
- the historical distribution of species (i.e. it must be geographically possible for a species to become established in a particular system)
- the other components of the biota (e.g. parasites, predators, food supply).

Aquatic communities are influenced by numerous natural and human-induced factors. Figures 2.1 and 2.2 emphasize the complexity of the various physical, chemical and biological factors that influence and determine the resultant biological community. Figure 2.1 divides these factors into chemical constituents and physical attributes, flow regime, habitat structure, energy source and biotic interactions. Figure 2.2 focuses on water-quality-related variables. From these diagrams it is clear that a combination of factors ultimately determines the composition of a biotic community. Given that many water quality variables differ in magnitude from region to region, it is reasonable to ask what the effects of these differences might be on aquatic organisms. Since there are so many variables that can affect the biota, only a few of the more obvious ones are dealt with below (Table 2.2). In addition to individual substances, aquatic ecosystems are often the recipients of effluents that contain a combination of chemical substances. Chapters on each of the major categories of substances, and on a number of whole effluents, are included in this review.

2.5 CATEGORIES OF WATER QUALITY VARIABLES

For convenience, variables can be grouped in a number of ways. Perhaps the simplest, and the system followed in this review, is to divide them into *physical* attributes such as temperature, turbidity and the concentration of suspended solids, and *chemical* constituents such as the total concentration of dissolved solids (TDS), and the concentrations of solutes such as gases and ions. Chemical variables can in turn be divided into *toxic* and *non-toxic* components. This is not entirely justified, in that virtually all chemical constituents can be shown to be toxic under certain circumstances, while some organisms are vastly more resistant to the toxic effects of particular chemical constituents than others are. For instance, the biota (including three species of flamingoes) of some high-altitude saline lakes in the South American Andes is able to survive apparently unharmed by arsenic concentrations of the order of 100 mg l^{-1} or more, far more than enough to kill flamingoes elsewhere in the world. It is nonetheless true that certain chemical constituents, such as trace metals, biocides and various other organic substances, have severe and sometimes lethal effects at relatively low concentrations. These constitute the toxic fraction.

2.6 SYNERGISM AND ANTAGONISM

Although each chemical substance has an effect, either beneficial or detrimental, on aquatic organisms, combinations of substances may be more, or less, toxic than each on its own. When two substances interact to produce a magnified effect, this is known as *synergism*. For instance, nickel and zinc are said to be synergistic in that they are more than five times more toxic in combination than either is alone. On the other hand, the toxicity of some substances is decreased in the presence of others, which are said to be *antagonists*. Calcium or magnesium, or high levels of alkalinity or TDS, for instance, reduce the toxicity of copper and other toxic trace metals. Changes in pH are particularly significant in altering the toxicity of a variety of chemical constituents, including trace metals. Organic compounds may reduce both acute and chronic toxicity of metals by complexing with the free metal ions.

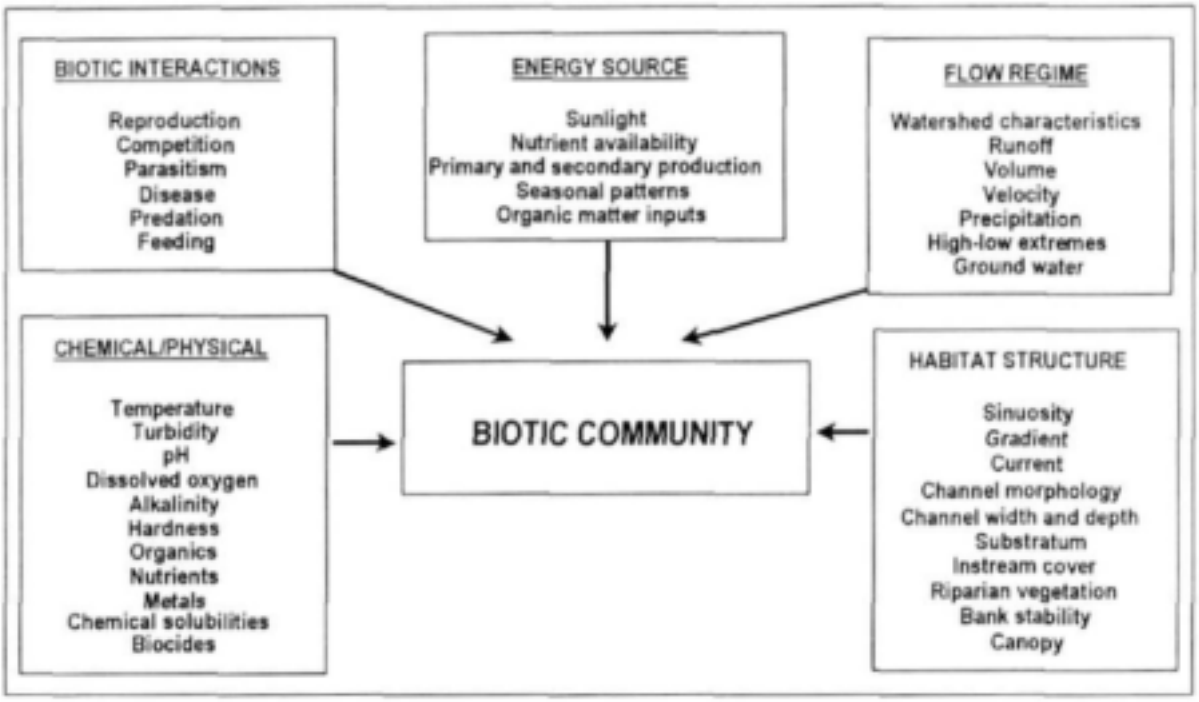


Figure 2.1 Some of the important chemical, physical and biological factors that influence and determine biotic communities [modified from Karr *et al.*(1986) cited by Rankin 1991].

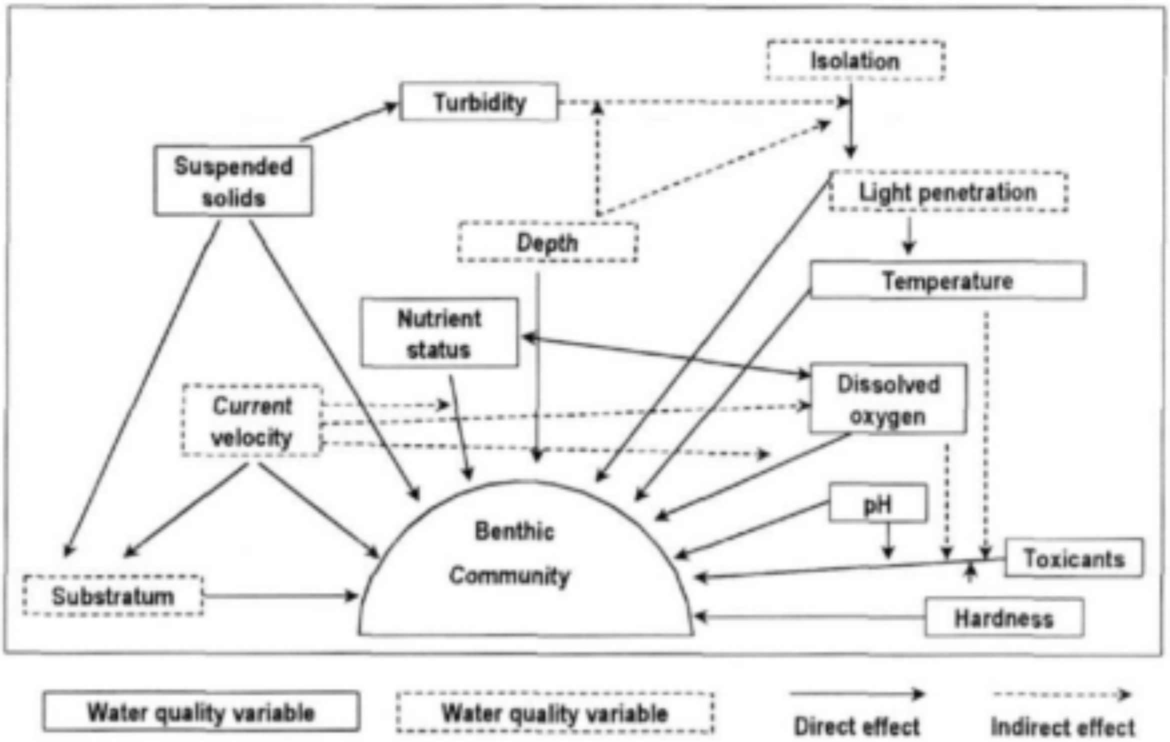


Figure 2.2 Major water quality and other variables potentially affecting aquatic biotas, showing both direct and indirect interactions between variables (modified from Hawkes 1979).

Table 2.2 The effects of some major physical attributes and chemical constituents of water in aquatic ecosystems. Whole effluents covered in this review are also listed.

WATER QUALITY VARIABLES	MAJOR EFFECTS	Chapter
PHYSICAL FACTORS		
Temperature	• Determines metabolic rate • Determines availability of nutrients and toxins • Determines oxygen saturation level • Changes provide cues for breeding, migration, etc.	4
Turbidity and suspended solids	• Turbidity determines degree of penetration of light, hence vision, photosynthesis. • Suspended solids reduce penetration of light, smother and clog surfaces (e.g. gills) and adsorb nutrients, toxins, etc.	5
CHEMICAL FACTORS		
pH	• Determines ionic balance • Affects chemical species and therefore availability • Affects gill functioning	6
Conductivity, salinity, TDS, individual ions	• Affect osmotic, ionic and water balance	7
Dissolved oxygen	• Required for aerobic respiration	8
Organic enrichment	• Reduces oxygen concentration • Increases nutrient levels	9
Nutrient enrichment	• Not toxic *per se*: cause eutrophication and thus affect community structure	10
Biocides	• Usually target specific groups (e.g. molluscs, insects, plants) and thus alter community structure	11
Trace metals	• Many essential at low concentrations • Some mutagenic, teratogenic, carcinogenic • Some metabolic inhibitors	12
TOTAL EFFLUENTS		
Agriculture		13
Forestry		14
Aquaculture		15
Engineering and construction		16
River regulation, environmental flows and inter-basin transfers		17
Industrial effluents		18
Mining		19
Urban runoff		20

2.7 TOLERANCE LIMITS AND THEIR SIGNIFICANCE

Over aeons of evolutionary time, succeeding generations of organisms of each species have become adapted to certain concentrations of each substance commonly found in the waters in which they live. The actual concentration range varies from species to species. For instance, the species of fish, and of other organisms, that live in mountain streams are adapted to TDS levels <200 mg l^{-1} and may well not be able to survive at values in excess of 500 mg l^{-1}. In contrast, those living in saline streams may be unable to survive at levels <50 000 mg l^{-1}, while marine fish live in a very narrow concentration range of 35 000 ±500 mg l^{-1}. Estuarine fish, on the other hand, may have to survive tidal cycles during which TDS varies between <1000 and 35 000 mg l^{-1} over a period of about twelve hours.

The concentration range over which a species can survive is known as the *tolerance range* and the upper and lower values are known as the *tolerance limits*. Within the tolerance limits, the *optimal range* is that to which organisms are most ideally suited and in which growth rates, fecundity and other measures of "health" are greatest. Organisms of a given species can survive on either side of this optimal range but as the tolerance limits are approached, more and more abnormalities become evident. The first signs are usually behavioural: for instance, fish may tend to avoid non-optimal conditions. Beyond this, physiological stress may become evident: respiratory, metabolic or excretory rates may increase, for instance. All of these changes may be accompanied by a decrease in egg and/or sperm production and hence in fecundity. Further, as tolerance limits are approached, organisms become more susceptible to parasites and pathogens and to food shortages. Very often adults are able to survive apparently unaffected by sub-optimal conditions, although breeding success may be greatly reduced if eggs cannot hatch or larvae cannot grow. Juvenile stages are frequently far more sensitive, however. It often happens, for instance, that an adult population of fish survives and grows under suboptimal conditions but there is no recruitment to the population because the juveniles do not hatch or grow. This is why juvenile stages are often used in toxicity tests (see Chapter 3).

2.8 "WATER QUALITY" AS A HUMAN CONSTRUCT

The term "water quality" was coined with reference to the quality of water required for human use: "good"-quality water is "pure" and unpolluted and suitable for drinking and stock watering, and hence *for agricultural and industrial purposes. It is critically important to acknowledge, however, that this is* entirely a human perspective. It so happens that most *freshwater* aquatic organisms might agree with this classification. It is emphatically not true for all aquatic organisms, however. Organisms living in the sea, in estuaries and in brackish or saline inland waters are unable to survive in water that is of "high quality" from a human perspective. When considering aquatic ecosystems, therefore, it is imperative that water quality is viewed from the perspective of the natural inhabitants rather than from a human perspective.

The corollary to these remarks is that one must be extremely wary of "improving" water quality to something approaching that acceptable to humans *if, in so doing, one is making it less tolerable for the natural inhabitants*. There is a fallacious tendency to assume that increasing the quantity of water in highly seasonal streams, or "increasing" the quality of water to human standards in naturally brackish wetlands, is in the interests of the biota. This *may* be so, but it is not *necessarily* so.

2.9 THE EFFECTS OF ALTERED WATER QUALITY ON AQUATIC ECOSYSTEMS

Human activities affect both the quantity and the quality of water in aquatic ecosystems. The effects of reductions in *quantity* of water are not discussed in detail in this volume. It should be noted, however, that such reductions sometimes interfere critically with the maintenance of natural ecosystem functioning, particularly during low-flow conditions when abstraction or damming result in increased evaporation and consequently in increased concentrations of chemical constituents in natural waters. Many aquatic ecosystems are affected by both reduced flows and increased levels of pollutants of various sorts. The linking of water quantity and water quality is discussed in Chapter 17.

Since each species has tolerance limits within which it can survive, variations in the concentration of chemical substances, and since these limits differ from species to species, alterations in water quality will affect different species to a greater or lesser extent. Greater and greater changes in water quality will thus gradually alter the constituent species of a biotic community until it is no longer recognisable as the same community. Some examples of this effect include:

- a shift in the physical position of a community of aquatic organisms (e.g. the community of organisms found below fish farms on mountain streams is more representative of communities naturally found much further downstream - e.g. Brown 1996)
- the introduction or loss of key species (e.g. the massive growth of algae in eutrophic waters may result in a "population explosion" of snails and the loss of mayfly and stonefly nymphs)
- reduction in diversity as a result of very small increases in the concentration of toxins such as trace metals
- reduced ecosystem functioning (e.g. the reduction and ultimately the absence of decomposition - and thus of nutrient cycling - in streams and lakes seriously affected by acid deposition).

In summary, the community of organisms living in a particular stream is determined by a number of physical and chemical attributes of the stream. One of the major determinands is water quality, which is frequently altered by human activities. The ways in which the effects of these alterations can be assessed are discussed in the following chapter.

CHAPTER 3: BIOLOGICAL ASSESSMENT OF THE EFFECTS OF WATER QUALITY ON AQUATIC ECOSYSTEMS

SUMMARY

Assessing the effects of changes in water quality and its constituents on aquatic ecosystems is complex. To produce any sensible guidelines for limiting the effects of reduced water quality on aquatic biotas, it is necessary to have some measure of biological integrity or ecosystem "health". Ecosystem health is quantified using what may be termed biological indicators. The use of biological indicators assumes that biotic components, such as algae, macroinvertebrates or fish, reflect the water quality conditions in which they live. Biological indicators may be individual species or whole communities. The science of ascertaining the effects of toxicants on individual species is generally referred to as toxicology. Toxicity testing, which is normally conducted in a controlled laboratory environment, allows the toxic effect of a constituent on an organism to be determined. A useful value is the concentration at which no effect is observed. Community-level analyses may test predictions based on single-species tests and may also predict indirect effects not measurable in single-species tests. An alternative approach to ecotoxicology is to measure actual biotic assemblages in the field environment. This process is termed bioassessment and it may be defined as the utilization of one or more components of the biota to assess the effect of a change in another component such as water quality. Measurement may take the form of structural components such as species composition, or functional aspects or processes such as the rate of decomposition. Several systems of bioassessment have been developed. Bioassessment is generally applied within the context of ecological reference conditions, which represent an expected, realistic and scientifically-authentic ecological benchmark against which bioassessment information is compared. Reference conditions facilitate data interpretation and allow the ecological significance of an effect to be established. It is necessary to be able to identify which observed differences in biotic assemblages stem from natural or intrinsic heterogeneity and variability in the system, and which are caused by anthropogenic activities such as reduced water quality.

3.1 INTRODUCTION

Some of the physical and chemical variables used to assess water quality for domestic, recreational, industrial and agricultural purposes include temperature, turbidity, total suspended solids (TSS), pH, conductivity, total dissolved solids (TDS), salinity, nutrient concentration, oxygen content, biological oxygen demand (BOD), chemical oxygen demand (COD) and toxins, including trace metals and biocides. The effects of changes in the magnitude of these variables are well known for domestic purposes and fairly well known for agricultural and industrial purposes and do not vary to any significant degree from region to region or from continent to continent.

Assessing the effects of changes in water quality and its constituent variables on aquatic ecosystems is far more complex. As we have shown in Chapter 2, each species of organism has specific tolerance limits for each variable, while each biotic community comprises a number of species of different, interacting, organisms. Further, tolerance limits of related species, or of isolated populations of widespread species, may vary geographically. This poses a problem, since the assessment of the effects of water quality on aquatic ecosystems and their inhabitants is a relatively new skill, and well-established norms are few. In South Africa, we have a dual problem: biotas adapted to a wide variety of conditions across the country; and very little knowledge about the responses of any of them to deterioration in water quality.

This chapter addresses the issues involved in assessing the effects of water quality on aquatic biotas generally and on South African biotas in particular. These issues, although complex, can essentially be reduced to two: determination of the best biological indicators for assessing water quality; and the ways of using this information to set guidelines for the management of water quality in South Africa's rivers: in other words, developing biological criteria for the protection of aquatic ecosystems.

3.2 BIOLOGICAL INTEGRITY OR ECOSYSTEM HEALTH

To produce any sensible guidelines for limiting the effects of reduced water quality on aquatic biotas, it is clearly necessary to have some measure of biological integrity or ecosystem "health". Biological integrity is required if an aquatic ecosystem is to support and maintain a balanced, integrated, adaptive assemblage of organisms having a species composition, diversity and functional organisation comparable to that of the natural habitats within a region (Karr & Dudley 1981). Ecosystem health, whilst being a useful and widely understood concept, is difficult to describe in precise, scientific terms (Schofield & Davies 1996). Implicit in the use of the term "health" is a societal judgement of wellbeing (e.g. Fairweather 1999) and, whilst there has been much criticism of the ecosystem health approach, it is widely used. Ecosystem health assessments require analysis of linkages between human pressures on ecosystems and landscapes, altered ecosystem structure and function, alteration in ecosystem services, and societal response (Rapport *et al.* 1998). Schofield & Davies (1996) understand ecosystem or river health to be a measure of the degree of similarity to a minimally-disturbed river of the same type, particularly in terms of its biological diversity and ecological functioning.

Ecosystem health is quantified using what may be termed *biological indicators*. Since the communities of organisms in any ecosystem are made up of *individual species*, it is reasonable to suppose that determining the responses of some of these species will provide an accurate reflection of the effect of changes in water quality on the ecosystems in which they live.

Changes in ecosystem health can be identified using either a "bottom-up" or a "top-down" approach (Scrimgeour & Wicklum 1996).

A "bottom-up" approach typically relies on data produced from simple laboratory systems, often at small temporal and spatial scales, to model changes in natural systems. It is often referred to as toxicity testing. Simply, toxicants can cause adverse effects on biological systems; toxicologists investigate these effects using toxicity tests, and ecotoxicology takes into account ecosystem effects (DWAF 2000). Short-term, single species toxicity tests are used to identify, regulate and monitor discharge of problem effluents into receiving ecosystems (Pontasch *et al.* 1989). More recently, studies have expanded from laboratory-based microcosm systems to experimental stream systems, called mesocosms, that allow toxicity assessments of whole communities (e.g. Guckert 1996). Applied aquatic toxicology is used in most developed countries as an important tool for the management of harmful chemical substances in aquatic ecosystems.

In the "top-down" approach, changes at the level of biotic assemblage and ecosystem are directly assessed in the natural environment, followed by identification of their causes. In ecosystems, two major categories of top-down endpoints are monitored, namely biotic structural components and ecosystem processes or functions. Characterising the response of an ecosystem to disturbance can be achieved using biological or ecological indicators and is termed instream biological response monitoring, or bioassessment (Roux *et al.* 1999). Bioassessment may be defined as the utilization of one or more components of the biota to assess the effect of a change in another component, such as water quality. It is the process of determining whether human activity has altered the biological properties of an ecosystem (Hawkins & Norris 2000). Potential components of the biota that may be used include protozoans (Joska *et al.* in press), periphyton (Barbour *et al.* 1999), macroinvertebrates (Reynoldson *et al.* 1997, Barbour *et al.* 1999, Dallas 2000a), fish (Karr 1981, Kleynhans 1999) and riparian vegetation (Kemper 1999). Bioassessment provides a time- and constituent-integrated assessment of the ecological or biological integrity of the system under consideration.

Clearly a range of biological indicators may be used to assess the effects of impaired water quality on aquatic ecosystems. The choice of indicator and the approach followed will depend on specific study and/or management objectives, and in some instance several different indicators may be needed for adequate assessment of the impact on an aquatic ecosystem. The range of biological indicators available is described below.

3.3 BIOLOGICAL INDICATORS

As mentioned, biological indicators of ecosystem disturbance may be individual species, which are commonly used in toxicological studies, or biotic assemblages. Biological indicators may also be functional and take the form of a process, such as rate of photosynthesis of decomposition. Some of the major biological indicators in each of these categories are listed in Table 3.1. There are advantages and disadvantages to using each of the types of biological indicator to assess the impacts of impaired water quality on aquatic ecosystems.

3.3.1 Toxicological studies

Measurements of growth rates, age to maturity, fecundity, etc., all provide information on the immediate effects of toxins or effluents on individual organisms. It is possible, if difficult, to carry out such investigations in the field if the organisms (e.g. fish) are large enough to be tagged, measured, released, re-caught and re-measured at appropriate intervals. Such measurements are valuable because they are performed on individual organisms that have been subjected to the totality of conditions in the stream from one measuring period to the next. The disadvantages are essentially two-fold. Firstly, although one can determine the effects of those conditions on the organisms being measured, one has very little idea of the actual conditions that caused the measured effects: any number of pulses of polluted water might have affected the organisms, but without continuous monitoring it is impossible to know how water quality was affected during these periods. Secondly, such techniques are only useful for large or sessile (attached) organisms. In practice such investigations are seldom achieved and most analyses of the effects of water quality on individuals are carried out in the laboratory. Indeed, huge numbers of laboratory studies have been carried out on the effects of individual water quality constituents on individual organisms of a variety of species. The process employed is termed "toxicity testing", whether or not the constituents or attributes to be tested are indeed toxic.

Table 3.1 Some biological indicators that can be used to estimate the effects of reduced water quality on aquatic ecosystems and their biotas.

ATTRIBUTES OF	BIOLOGICAL INDICATOR	COMMENT
Individual species	Behaviour Growth rate Metabolic rate Sensitivity to pathogens Condition Fecundity Age to maturity Survival rate Abundance Biomass Recruitment and turnover	Used in toxicity testing (usually laboratory-based)
Biotic communities	Species composition Biodiversity (e.g. number of species) Complexity of interrelationships Community succession Alteration in key species Resilience to change Sensitivity to change	Field data and community-level toxicity tests
Natural processes	Rate of photosynthesis Rate of nutrient cycling Rate of decomposition	Field data

Toxicity tests may be conducted using standard laboratory organisms such as water fleas (*Daphnia*) and guppies *(Poecilia reticulata)* or using indigenous organisms from rivers and may be based on one or more species. Toxicants tested may be single substances or complex mixtures. *Regardless of the approach followed, the results of toxicity tests link chemical concentrations to* biotic responses, providing an interpretive link between information on water chemistry (provided by routine chemical monitoring) and information on ecosystem health (provided by biomonitoring) (DWAF 2000).

3.3.1.1 Single-species toxicity tests

Procedures for determining the toxic effects of any constituent or attribute of water on living organisms have to be extremely carefully carried out. The reader is referred to Butler (1978) and Hellawell (1986) for details. In brief, the major considerations are the organisms used, the procedures carried out, the statistical analysis of the results, and the use of the results for management purposes. In order to obtain reliable (and therefore repeatable) results, the biology of the test organisms must be well known. For instance, if the test is to examine the effects of a toxin on any aspect of an organism's life history, then its growth rate, age to maturity, number of eggs produced, etc., must be known for a series of standard conditions. All species are intrinsically variable genetically, and that variability varies from population to population and from time to time. Thus to allow results to be compared from laboratory to laboratory, or from one toxin to the next, standard laboratory strains are maintained for testing purposes. The cost of developing "life tables" for even a single species is considerable.

Most testing is done on internationally standardized strains of very few species. The water flea *Daphnia magna* is the guinea-pig of the water testing laboratory, being small and reproducing both rapidly and asexually. A number of other invertebrates such as *Artemia salina* (the brine shrimp), *Gammarus* spp. (amphipods), *Tubifex* spp. (oligochaete worms), chironomid larvae and mayfly nymphs are also used on occasion, although their biology is much more poorly known. Commonly used algae include species of *Chlamydomonas, Scenedesmus* and *Microcystis*. Fish commonly used in the northern hemisphere include various species of trout and salmon as well as guppies *(Poecilia reticulata)*, fathead minnows and bluegill sunfish.

In South Africa in the 1980s, the Hydrological Research Institute (now called the Institute for Water Quality Studies – IWQS) investigated the feasibility of using toxicity testing as a component of surveillance monitoring (Roux 1990). Roux recommended that such monitoring involve tests with at least three species, preferably a bacterium (*Pseudomonas putida*) or a cyanobacterium (*Microcystis aeruginosa*), an alga (*Selenastrum capricornutum*) and a daphniid (*Daphnia pulex*). Toxicity testing procedures were developed using bacteria (Slabbert 1986; Slabbert & Grabow 1986; Slabbert 1988; Truter 1991a), protozoans (Slabbert & Morgan 1982; Slabbert *et al.* 1983; Slabbert & Maree 1986; Truter 1991b), *Daphnia* (Truter 1990) and fish (*Poecilia reticulata*: Truter 1991c). A brief review is provided by Grabow *et al.* (1985).

Recently, research into the use of indigenous riverine organisms in toxicity testing has been undertaken in South Africa (Scherman & Palmer 2000). A key motivation for this research was the fact that most aquatic ecosystems in South Africa are rivers and it is thus necessary to develop toxicity tests using insect larvae and other riverine invertebrates. These organisms are also useful for testing the effects of chemicals and mixtures in the field (DWAF 2000), thus contributing to site-specific water quality guidelines and to the refinement of South African guidelines for aquatic ecosystems. Other taxa may also be useful test organisms. For instance Channing (1998) has conducted a baseline study of the value of tadpoles as bio-indicators of the effects of agrochemicals, and Joska *et al.* (in press) have investigated the use of protozoans in biomonitoring of water quality.

In relation to using indigenous riverine organisms in toxicity testing, a facility has been developed for producing standard laboratory organisms for ecotoxicological research (Haige & Davies-Coleman 1999). The need to have an understanding of the physiology and functioning of test animals under natural conditions, so that functional impairment due to contaminants and toxins can be assessed, prompted in-depth research into the life history, biology and ecology of selected test organisms. This research, together with that of Palmer *et al.* (1996) and Scherman *et al.* (2003), have made a significant contribution to the field of ecotoxicology in South Africa.

3.3.1.2 Whole-effluent toxicity (WET) tests

Whole-effluent toxicity tests have been widely used to assess the potential effects of effluent discharges on aquatic ecosystems. Slabbert *et al.* (1998) have developed procedures to assess Whole Effluent Toxicity (WET) tests, and guidelines for WET testing, in South Africa. A recent review (La Point & Waller 2000) examined the linkages between WET and associated field biomonitoring. According to these authors, previously published results indicated that standards for the usual WET freshwater test species, namely *Ceriodaphnia dubia* and *Pimephales promelas*, may not always protect most of the species inhabiting a stream receiving a polluting effluent. La Point & Waller conclude that WET testing does not address bioconcentration or bioaccumulation of hydrophobic compounds, does not assess eutrophication effects in receiving streams, does not reflect genotoxic effects, and does not test for endocrine-disrupting chemicals.

3.3.1.3 Toxicity assessment by community analyses

Community-level analyses may be used to test the predictions derived from single-species tests and to determine potential indirect effects (Guckert 1996). Often mesocosms (artificial outdoor or indoor streams) are used to ascertain ecosystem-level NOECs (no-observed-effect concentrations) of toxicants. In contrast, multispecies tests in laboratory microcosms provide more specific predictions and increase the level of confidence in estimates of permissible concentrations by increasing the range of organisms examined (Pontasch *et al.* 1989).

3.3.1.4 Measuring toxic effects

The effects of a toxin on test organisms can be measured in a variety of ways. The most final is, of course, the death of the organism. But other common measures include effects on growth rate, egg production, oxygen consumption, chlorophyll production, photosynthetic rate, etc. The No-Observed-Effect-Concentration (NOEC) is also used, particularly in the United States. Constituent-specific water quality criteria developed in South Africa for toxic constituents (DWAF 1996) use the Chronic Effect Value (CEV) and the Acute Effect Value (AEV) as biocritera. The CEV is defined as that concentration or level of a constituent at which there is expected to be a significant probability of measurable chronic effects in up to 5% of the species in the aquatic community. The AEV is defined as that concentration or level of a constituent at which there is expected to be a significant probability of acute toxic effects in up to 5% of the species in the aquatic community. Toxicity data may be expressed in many different ways (Table 3.2). Results are seldom clear-cut, particularly at concentrations that have minimal effects, and so very specific statistical analyses are used to determine the final concentration at which an effect is considered to have taken place.

Table 3.2. Terms and abbreviations used in expressing toxicity data (modified from Hellawell 1986).

TERM	ABBREVIATION	DEFINITION
No-observed-Effect-Concentration	NOEC	The concentration at which no effects are observed
Chronic Effect Value	CEV	The concentration at which there is expected to be a significant probability of measurable chronic effects in up to 5% of the species in the aquatic community
Acute Effect Value	AEV	The concentration at which there is expected to be a significant probability of acute toxic effects in up to 5% of the species in the aquatic community
Median effective concentration	EC_{50}	Concentration at which a specified effect is observed in half of the test population
Median lethal concentration	LC_{50}	Concentration at which half the test population dies
Median tolerance limit	TL_M	Concentration at which half the test population dies
Median effective time	ET_{50}	Time taken for an observed effect to occur in half of the test population
Median survival time	LT_{50}	Time taken for half of the test population to die
Incipient lethal level	ILL	Highest concentration that organisms can tolerate indefinitely

3.3.1.5 Advantages and disadvantages of toxicity tests

An advantage of strictly controlled toxicity testing is repeatability. However, single-species toxicity tests are considered to suffer from low environmental realism, although they can be effective if the underlying mechanisms of environmental change are known (Scrimgeour & Wicklum 1996). The ability of single-species tests to protect entire ecosystems has also been questioned, and thus micro/mesocosm tests have been proposed as additional tools for reducing uncertainty in ecological risk assessment (e.g. Pontasch *et al.* 1989, Guckert 1996). The shortcomings of single-species tests have been summarised by De Vlaming *et al.* (2000). They suggest that

"single-species tests do not characterise the persistence/duration or frequency of exposures in ambient waters without repeated sampling and testing; they do not directly measure biotic community responses; they do not encompass the range of species sensitivities or function responsive to toxic chemicals that occur in biological communities; neither delayed impacts or effects due to bioaccumulation and bioconcentration are measured; the highly controlled exposure regimes in these laboratory tests do not reflect the multivariate and complex exposure conditions in aquatic ecosystems; results probably underestimate biotic community responses to chemicals because of multiple stressors acting on aquatic ecosystems; and use of non-resident species may not represent toxicological sensitivities in some aquatic ecosystems".

Recent advances in toxicity testing, through the adoption of multi-species testing, the use of micro- and mesocosms, the use of indigenous organisms, and the examination site-specific conditions, have been instituted with the intention of making toxicity testing becoming more useful in the management of aquatic ecosystems.

3.3.2 Communities as biological indicators: bioassessment

Whilst toxicity testing provides useful information, it is advantageous to have some means of determining the effects of changes in water quality on whole ecosystems. A method, preferably numerical, is needed to estimate biological integrity and ecosystem functioning at any given time.

Biological monitoring or bioassessment procedures attempt to assess the effects of changes (either improvements or degradations) in water quality on aquatic ecosystems. Bioassessment has been acclaimed as a more sensitive and reliable measure of environmental conditions than either physical or chemical measurements (Warren 1971) and using living organisms as indicators of disturbance in an ecosystem has proven successful (Rosenberg & Resh 1993).

3.3.2.1 Bioassessment (or biomonitoring) versus physico-chemical monitoring

Traditionally, physico-chemical monitoring formed the backbone of water quality monitoring in South Africa (DWA 1986) and elsewhere (e.g. Barbour *et al.* 1996), and control of surface water quality has been through the control of effluent discharges. Assessment of the common physical attributes and chemical constituents of water, although essential for determining the type and concentration of pollutants entering a river, is limited to the period of sample collection and to the physical and chemical analyses performed.

Widely recognised limitations of physico-chemical monitoring include the intermittent nature of the measurements, in that unless samples are collected continuously, pulsed releases of effluents that result in an alteration of water quality may not be recorded. Furthermore, the potential number of constituents that could be present is vast, while routine analyses are usually limited to non-toxic determinants such as temperature, conductivity, total alkalinity and nutrient concentrations. The number and variety of potentially toxic compounds (e.g. trace metals, biocides) that could affect water quality is considerable, as is the cost of analysing the full range of these compounds, and routine testing for all possible toxins is thus unrealistic. The sensitivity of chemical analytical methods when measuring very low concentrations of pollutants may also be inadequate, particularly for substances that are characteristically present in these low concentrations but which are persistent and tend to accumulate in the environment.

A further complicating factor, when assessing the effect of altered water quality by means of physical and/or chemical data, is that of synergism and antagonism. Although each water quality variable has an effect on aquatic organisms (beneficial or detrimental), the overall effects of changes in the magnitude of more than one variable may be greater or less than the effect of each in isolation. For example, changes in pH are particularly significant in altering the toxicity of a variety of chemical constituents, including trace metals. These subtle magnifying and reducing effects would not necessarily be revealed by routine physico-chemical monitoring.

While traditional physical and chemical evaluations of water quality have largely been inadequate (Warren 1971, Barbour *et al.* 1996), as has the use of physical and chemical standards to protect the aquatic environment from, for example, downstream effects of wastewater treatment works in South Africa (Dickens & Graham 1998). Further, while focusing on physico-chemical monitoring, other structural impacts that have led to alterations of river flow, loss of habitat area, loss of habitat diversity, obstructions to passage through streams and riparian degradation, have also been overlooked (Harris 1995 cited by Schofield & Davies 1996). Because they are dependent on the medium in which they live, aquatic organisms are sensitive to all alterations to the water body by, for example, pollution or habitat alteration, so such alterations will, of course, be reflected in the composition of the biotic assemblages found in the system. The biota therefore act as indicators of the overall ecological condition of the aquatic ecosystem, by acting as continuous monitors of the water they inhabit (Hawkes 1979), thereby enabling long-term analysis of both regular and intermittent discharges, variable concentrations of pollutants, single and multiple pollutants, and synergistic or antagonistic effects.

Whilst indicating that a water body is impacted, living organisms seldom provide insight into the cause of the problem. For this reason, Hawkes (1997) suggests that bioassessment, which produces biological data, and physico-chemical monitoring, which produces physical and chemical data, are complementary. He suggests it would not be useful to correlate the two assessments, as bioassessment information is probably of greatest value when it does not confirm the chemical data, thus revealing the effect of other physical or chemical factors. Reynoldson & Metcalfe-Smith (1992) suggest that biological systems should be the standard for monitoring, assessment, and target formulation, and that the role of chemistry and physics is most important in the identification of factors causing impairment and the selection of appropriate remedial actions.

3.3.2.2 Using macroinvertebrates in bioassessment

There is general consensus that benthic macroinvertebrates are amongst the most sensitive components of aquatic ecosystems and they have been widely used in bioassessment. Briefly, as summarised by Rosenberg & Resh (1993), macroinvertebrates are ubiquitous and diverse, and are therefore affected by a variety of disturbances in many different types of aquatic habitats. Sensitivity to stress varies with species and the large number of species within an assemblage thus offers a spectrum of responses to environmental stresses. In their aquatic phase, macroinvertebrates are largely non-mobile and are thus representative of the location being sampled, which allows effective spatial analyses of disturbance. They have relatively long life cycles (compared to other groups such as planktonic organisms) allows elucidation of temporal changes caused by disturbances. A major limitation in the use of macroinvertebrates in bioassessment is their heterogeneous distribution and patchiness that result in natural spatial and temporal variability in macroinvertebrate assemblages (e.g. Marchant 1988, Palmer *et al.* 1991, Dallas 2002b).

3.4 BIOASSESSMENT SYSTEMS

The following section provides an overview of the historical and current methods used in bioassessment. Most methods make use of the taxonomic composition of biotic systems. It should be noted, however, that no one system provides a complete quantitative analysis of all of the organisms present in a river, but that each attempts to relate certain facets of ecosystem structure or function to the overall biological integrity or functioning of the system. Details of the earlier methods are available in Hawkes (1979), Metcalfe-Smith (1991) and Reynoldson & Metcalfe-Smith (1992).

3.4.1 The 'Saprobian' System

Kolkwitz and Marsson (1908, 1909 cited by Warren 1971) first used the presence and absence of different organisms to develop their "saprobic" system of zones in organically polluted rivers. In these studies, emphasis was placed on the relative tolerance levels of individual species of organisms and hence on the presence of certain species. Taxonomic effort was great (i.e. all specimens were identified to species) and all trophic levels were examined. The original system was geographically specific (Germany) and was only appropriate for sewage pollution (Chutter 1972).

3.4.2 Diversity indices

Diversity indices are mathematical expressions of three components of community structure, namely richness (number of species present), evenness (uniformity in the distribution of individuals among species) and abundance (total number of organisms present) that can be used to describe the response of a community to the quality of its environment. The earliest forms of diversity indices (Margalef 1951 cited by Warren 1971) expressed the species richness of a community as the number of species relative to the total number of individuals. A modified version (the Shannon-Weiner index), which is now commonly used, also takes into account the number of individuals per species (Wilhm & Dorris 1968 cited by Hawkes 1979). Use of this index in pollution studies is based on the premise that species diversity decreases as human-induced ecological stress increases. This is not always true, however, as low diversity may be caused by physical stresses, for instance in torrential mountain streams which nonetheless have excellent water quality (Wells 1992) and in cases where pollution may be masked by a change in species composition (from sensitive to tolerant species) that would not be reflected in any species diversity index. Intensive sampling, sorting and identification are required to use these methods effectively. As a result, comparisons between studies are seldom statistically valid; Boyle *et al.* (1990), for instance, were of the opinion that diversity indices are not appropriate indicators of ecosystem integrity unless they are used together with other indices.

3.4.3 Biotic indices

Biotic indices are numerical indices that use one or more components of the biota to provide a measure of the biological condition of a site. One of the advantages of biotic indices is that they formalise what any good biologist, familiar with local biota, knows about the biological condition of a stream and they communicate biological condition to policy makers and concerned citizens, thus providing a scientific basis for management decisions that affect aquatic resources (Fore *et al.* 1996). Biotic indices based on macroinvertebrate assemblages have proved to be useful measures of stream or ecosystem "health" and are widely applied today (Hellawell 1986, Rosenberg & Resh 1993), with many countries beginning to rely on biological assessments as their primary measure of the ecological health of surface waters (Gerritsen *et al.* 2000). Historically, biotic indices have often been calculated *a posteriori* from quantitative macroinvertebrate sampling (e.g. Chutter 1972, Hilsenhoff 1988). Labour and time constraints associated with such quantitative sampling have prompted the development of qualitative rapid bioassessment methods such as the BMWP system (Biological Monitoring Working Party, e.g. Wright 1995), the Australian SIGNAL biotic index (Stream Invertebrate Grade Number Average Level, Chessman 1995) and SASS (South African Scoring System, Chutter 1998). Not only do these rapid bioassessment methods utilise simplified data interpretation methods *via* the generation of biotic indices, but several also reduce the time needed to process samples, either by being field-based (e.g. SASS), by limiting the number or organisms identified (i.e. fixed-count method such as SIGNAL), or by limiting taxonomic resolution to that of family or higher (e.g. SASS, SIGNAL).

Whilst there is still much debate about the potential loss of information that may occur when biotic indices are used (Brown 1997), for example by omitting abundances from the index calculation for SASS, they have been used to reveal the effects of many different anthropogenic impacts. Biotic effects on riverine macroinvertebrate assemblages that have been effectively assessed using biotic indices include the effects on receiving water bodies of organic pollution (Cao *et al.* 1997) via discharges from sewage treatment works (e.g. Chessman 1994, Wright *et al.* 1995), wastewater discharges (e.g. Chessman *et al.* 1997, Dickens & Graham 1998) and trout farm effluent (e.g. Loch *et al.* 1996, Brown 1997), the effects of mixed diffuse runoff such as urban storm water runoff (e.g. Chessman *et al.* 1997), the effects of agriculture (e.g. Quinn *et al.* 1997), afforestation (e.g. Quinn *et al.* 1997, Rothrock *et al.* 1998), metal pollution (e.g. Carlisle & Clements 1999) and experimental insecticide treatments (Wallace *et al.* 1996).

In South Africa, aquatic macroinvertebrates are one of the most commonly assessed components of the biota and SASS is used as the routine rapid bioassessment tool to assess water quality and general condition of riverine ecosystems. It forms the backbone of the River Health Programme (RHP), a national programme aimed at assessing the ecological state of aquatic ecosystems in South Africa (Dallas 2000a, Dickens & Graham 2002). Briefly, SASS is a scoring system based on macroinvertebrates, whereby each macroinvertebrate taxon is allocated a sensitivity/tolerance score according to the water quality conditions it is known to tolerate (Dallas *et al.* 1995, Dallas 1997). Data interpretation is based on two calculated values, namely SASS Score, which is the sum of the sensitivity/tolerance scores for taxa present at a site, and average score per taxon (ASPT), which is the SASS Score divided by the number of taxa. SASS has proved to be an efficient and effective means of assessing water quality impairment and general river health (Dallas 1997, Chutter 1998).

3.4.4 Community Comparison indices

Even when frequent, complete sampling of macroinvertebrates or other organisms is undertaken, analysis of the data is problematical. A wide range of computer-based similarity and dissimilarity indices, all intended to provide comparisons between sites or between sampling programmes, are currently in use. They all have somewhat different mathematical properties and thus analyse different properties of the ecosystems under consideration (Reynoldson & Metcalfe-Smith 1992). Because of these differences, each of the programs may generate different results when applied to the same data set. For this reason, the objectives of the analysis, and type of data available, must be carefully considered before community comparison indices are applied. The Bray-Curtis index has been identified by Hruby (1987 cited by Reynoldson & Metcalfe-Smith 1992) as the most valuable because of the following properties: "Species abundances are included; the index uses transformed data, which increases the importance of rare species; the index values are not clumped; and there is a linear response to changes in species numbers and abundances". Boyle *et al.* (1990) tested the sensitivity of this index, however, and found that it was relatively insensitive at low levels of perturbation. Essentially, similarity indices compare community assemblages at different sites and/or at different times. These methods are informative and less reliant on sampling intensity, but require realistically comparable sites, sampling and analytical methods, and taxonomic effort.

3.4.5 Reduced Assemblages

The users of reduced assemblages assume that useful information about the integrity of a biotic community can be gleaned by examining a single well-known component of that community. In terms of macroinvertebrates, commonly used groups include oligochaetes, chironomids and trichopterans. All three groups have been shown to possess species with a wide variety of tolerance levels to different types of pollution: i.e. individual species within each group are variably tolerant of different types of pollution. By limiting the assessment to one group, the taxonomic refinement within the group can be greater and individual research workers are able to use their expert judgement in relation to the group with which they are most familiar.

3.4.6 Functional Feeding Groups (FFGs)

Macroinvertebrates can be classified into FFGs according to their feeding mechanisms. Common FFGs include scrapers, grazers, collectors, shredders, predators and deposit feeders. In a natural system, FFGs would occur in predictable proportions, determined by the natural longitudinal changes that occur in food sources down the length of a river or stream. It is assumed that any disturbance would disrupt this pattern. The method requires identification to species and a good knowledge of the feeding mechanism of each species. Since this approach is based on nutrient dynamics it can really be used to assess the effects of organic enrichment only (Metcalfe-Smith 1991). Furthermore, the information available on the FFGs of South African riverine organisms is poor; there is also some doubt about the theoretical basis of studies of this kind. The topic is reviewed in Palmer (1991).

3.5 DATA INTERPRETATION AND ECOLOGICAL SIGNIFICANCE

Given that *the ultimate objective of bioassessment is to evaluate the effect of anthropogenic activities on biological resources*, it is necessary to be able to identify those differences in the biotic assemblage that stem from natural or intrinsic heterogeneity and variability in the system, and those that are caused by anthropogenic activities such as reduced water quality. This requires an understanding of the natural or expected conditions, with respect to the component of the biota or ecosystem being examined.

An understanding of what constitutes an "ecological significant effect" applies equally to toxicity tests (particularly micro- and mesocosm studies and whole-effluent toxicity tests) and to biological assessments. The ecological significance of a particular result depends on a number of factors, including the magnitude, spatial and temporal extent, recovery or reversal of the event, and the natural variability inherent in the measured endpoint (La Point & Waller 2000).

Tools for interpreting bioassessment data, such that an observed effect is in some way quantified, vary from comparatively simple tables that provide values for different categories of impact (e.g. Chutter 1998) to complex predictive models, which relate environmental variables to biotic communities (e.g. Wright 1995, Smith *et al.* 1999). Whatever level of complexity is adopted in data interpretation, it is necessary to know the "expected" condition, either as an expected biotic index value or as an expected macroinvertebrate assemblage or both. This "expected" condition is referred to as the reference condition. Bioassessment is generally applied within the context of ecological reference conditions, which represent an expected, realistic and scientifically-authentic ecological benchmark with which bioassessment information is compared.

3.5.1 Ecological reference conditions

An ecological reference condition is the condition that is representative of a group of minimally-disturbed or "least-impacted" sites organised by selected physical, chemical and biological characteristics (Reynoldson *et al.* 1997). Reference conditions enable the degree of degradation or deviation from natural conditions to be ascertained, and thereby serve as a foundation for developing biological criteria for the protection of aquatic ecosystems.

A reference condition is usually derived from a suite of similar reference sites, termed a regional reference condition, although single site-specific reference conditions are sometimes also used. Site-specific conditions are typically used in an upstream/downstream or "paired" scenario where a monitoring site is compared to the condition at a single reference site, and are needed when there are concerns with specific point sources. A typical example of a "paired scenario" would be sites upstream and downstream of the point of discharge from a waste-water treatment plant.

Regional reference conditions are necessary because pristine sites, particularly in the lower reaches of rivers, generally no longer exist, and near-pristine sites are often scarce. Inferences need to be made from minimally-disturbed sites to those impacted by human activity. Approaches for deriving ecological reference conditions for riverine macroinvertebrates have been well documented (Dallas 2000a, b; Dallas 2002a, b; Reynoldson & Wright 2000).

3.5.2 Interpreting bioassessment data using ecological reference conditions

Management action depends on the knowledge that a certain impact causes an aquatic assemblage or ecosystem to respond in some way that is outside the natural range of variation (Roux *et al.* 1999) and the ultimate objective of any bioassessment programme is to facilitate the detection of disturbance at a site as reflected by one or more components of the biota. Reference conditions facilitate this by defining what is expected at a site and provide a means of comparing observed conditions with expected conditions. This is a complex task and one that requires careful consideration of factors that may potentially affect data interpretation. Any reference condition is also likely to be a dynamic one, changing as our ecological understanding of the system grows (Meyer 1997).

Guidelines or biocriteria for the interpretation of environmental conditions with respect to established reference conditions have been formulated by several authors (e.g. Hughes 1995, Minns 1995) and, as already mentioned, range from simple interpretative tables to more complex predictive models. Minns (1995) proposed the following rule: "If any ecosystem is to retain the inherent capacity to return to its original state given the removal of all human alterations and stresses, any degree of change greater than 50% relative to the original state, is unlikely to be tolerated". Similarly, Hughes (1995) suggests that: (1) 90% of the reference condition is still high quality and perhaps within the range of natural and measurable variability, (2) 75% of the reference condition is still acceptable, (3) 50% - 75% of the reference condition could be considered marginal, and (4) less than 50% of the reference condition is unacceptable. These ranges are not based on actual data, but some sliding of this kind is no doubt a fair reflection of reality; the extent to which the percentages proposed by Hughes (1995) reflect the degree of impairment is a moot point.

In predictive modeling systems such as RIVPACS (Wright *et al.* 1993, Wright 1995) and AusRivAS (Furse 2000, Simpson & Norris 2000), the use of "biological banding" systems with different bands representing different biological conditions, serves to simplify data interpretation and to aid management decisions (e.g. dallas 2002a, b). The severity of any environmental impact is assessed based on how much the observed (O) parameter deviates from the expected (E), i.e. reference condition.

CHAPTER 4: TEMPERATURE

SUMMARY

The thermal characteristics of running waters are dependent on various hydrological, climatic and structural features of the region, catchment area and river. Running waters in regions of seasonal climates exhibit daily and seasonal temperature patterns, in addition to longitudinal changes along a river course. All organisms have a temperature or range of temperatures at which optimal growth, reproduction and general fitness occur. Changing water temperature may expose aquatic organisms to potentially lethal or sublethal conditions. Anthropogenic causes of temperature changes in river systems include those resulting from thermal pollution, stream regulation and changes in riparian vegetation. An increase in water temperature decreases oxygen solubility and may also increase the toxicity of certain chemicals, both which result in increased stress in the associated organisms. Many life cycle characteristics of aquatic organisms are cued into temperature, i.e. temperature is the cue for migration, breeding, emergence, etc. Temperature changes affect metabolic processes and life cycle patterns by altering reproductive periods, rates of development and emergence times of aquatic organisms. Differences in temperature tolerance amongst the biota, and regional and seasonal temperature differences, should be considered when establishing guidelines for the management of water temperature in rivers.

4.1 INTRODUCTION

Natural thermal characteristics of running waters are dependent on hydrological, climatological and structural features of the region and catchment area (Table 4.1). Hydrologically, factors such as the source of water (snow melt, surface runoff, lake outlet, etc), the relative contribution of ground water, and the rate of flow or discharge, will influence the temperature regime (Ward 1985). The latitude and altitude of the river, as well as climatic factors such as air temperature, cloud cover, wind speed, vapour pressure and precipitation events, all influence the thermal conditions in rivers. For example, Appleton (1976) recorded large, nearly instantaneous depressions of stream temperatures caused by summer hailstorms in an eastern Transvaal stream in South Africa. Structural characteristics of the river and catchment area, including topographic features, vegetation cover, channel form, water volume, depth and turbidity, affect the amount of solar radiation reaching and heating the water, and thus its thermal regime. The high specific heat of water results in an extremely high capacity for heat absorption and extremely slow release of heat and thus contributes to thermal stability, which in turn has important consequences for aquatic species (Reid & Wood 1976).

Temporally, running waters in regions of seasonal climates exhibit daily and annual temperature periodicity, in addition to longitudinal changes along a river course. The minimum and maximum temperatures, and temperature ranges, vary depending on the factors mentioned above. Except for birds and mammals, all organisms associated with fresh water are poikilothermic: in other words, they are unable to control their body temperatures, which are therefore the same as that of the ambient water. These animals and plants are very susceptible to changes in water temperature since a 10°C increase results in a doubling of the organism's metabolic rate (Hellawell 1986). Many organisms are adapted so that seasonal changes in temperature act as cues for the timing of migration, spawning or emergence. For each species there is a temperature at which optimal growth, reproduction and general fitness occur, as well as a range in which normal activities can continue. The maximum and minimum points on this range are the lethal limits.

Changes in water temperature that are unrelated to natural variations may have an effect at the *organism, species and/or community level. Sudden temperature changes, even if relatively small,* may have a significant effect, i.e. the rate of change, as well as the absolute new value, may be important. Howells (1983, p.11) summarized all the possible effects a temperature change may have:

> "At the organism level, excessive temperature may in the extreme result in death, but at lower ranges may influence movement, behaviour, the onset or maintenance of sexual maturity, life stage development, and growth and size. At the species level effects manifest as adaptation or genetic selection, whilst in communities, temperature may influence intra-specific selection, predator-prey interactions, or other intra-specific interactions. Hence it is to be concluded that ecological effects might include changes in population abundance, and diversity, in addition to standing crop and productivity."

Table 4.1 Major factors affecting the thermal regime of rivers.

FEATURE	FACTOR
Hydrological	Source of water (snow melt, surface runoff, lake outlet)
	Groundwater contribution
	Flow rate and discharge
Climatological	Latitude and altitude of the river
	Air temperature
	Cloud cover
	Wind speed
	Vapour pressure
	Precipitation events
Structural	Catchment and river topography
	Vegetation cover
	Channel form
	Water volume, depth
	Turbidity

4.2 ANTHROPOGENIC CAUSES OF WATER TEMPERATURE CHANGE

A change in water temperature may be either an increase or decrease, although it is more often increases that occur as a result of anthropogenic activity and that are therefore better documented in the literature.

4.2.1 Thermal pollution

The introduction of heated effluent (including heated industrial discharge, heated cooling waters from power stations and returning irrigation waters) into a natural water body results in thermal pollution.

4.2.1.1 Heated power station discharges

The effect of heated effluent discharged from power stations on aquatic ecosystems is variable and, to some extent, dependent on season (Wellborn & Robinson 1996) and on the degree to which the heated effluent mixes with the receiving water (Mann 1965). Several studies have reported a deleterious effect on aquatic biotas (e.g. Mann 1965, Wellborn & Robinson 1996). Mann (1965) concluded that the critical aspect of heated discharges was related to the pattern of discharge. The most deleterious effect occurred when a surface layer of heated water covered the whole river, as this "concentrated" the temperature increase and prevented organisms that periodically surface from doing so comfortably. Other potential patterns included a narrow band of hot water on top of a cold layer (a common pattern in fast-flowing narrow streams) and the complete horizontal mixing of the discharge water with the river water, which would normally result in an overall temperature increase of only a few degrees and hence in a lesser impact.

4.2.1.2 Heated industrial discharges

Heated industrial discharges are frequently linked to other forms of pollution (e.g. chemical pollution). No published studies have isolated the effect of the temperature changes *per se*, although presumably the effects of elevated temperature would be similar to those of power-station discharges. The toxicity of the associated chemicals may be significantly increased when released in association with elevated temperatures (Duffus 1980, Förstner & Wittmann 1981).

4.2.1.3 Returning irrigation water

Stream temperatures may increase by 10 to 20°C as a result of irrigation practices and the return of agricultural drainage (Eldridge 1960 cited by McKee & Wolf 1963). No specific published studies on the consequences for stream biotas were found during this literature survey, presumably because returned irrigation water is not a 'point' source and thus the effects are difficult to isolate.

4.2.2 Stream regulation

The extent to which an upstream impoundment modifies downstream thermal conditions depends on operational variables (release depth, discharge pattern), limnological variables (retention times, stratification pattern and thermal gradients), and the position of the impoundment along the longitudinal profile of the river (Ward & Stanford 1982). Deep-release dams typically result in summer-cool and winter-warm downstream conditions (Pitchford & Visser 1975). Nowadays, engineers and associated developers responsible for the construction of water storage reservoirs are aware of the importance of maintaining the natural temperature regime below an impoundment, and often make provision for this by designing structures and management regimes that facilitate the ecologically sensible release of water (e.g. multi-level outflows that mimic natural changes in discharge). Many small farm dams (< 250 000 m^3) constructed on small tributaries do not have any of the above control measures and potential temperature impacts may be significant, particularly since they are small and thus heat up rapidly and do not stratify.

4.2.3 Changes in riparian vegetation

On removing riparian vegetation that provides shading to a river, water is exposed to increased direct solar radiation, which leads to higher temperatures and greater temperature ranges and fluctuations. Graynorth (1979) compared the thermal characteristics of a stream flowing through undisturbed evergreen beech forest (South Island, New Zealand) with one whose banks had been clear-felled. The latter was cooler in winter (mean ≈ 1.5°C), warmer in summer (mean ≈ 6.5°C) and had greater temperature fluctuations (mean ≈ 9.4°C). Canopy-like riparian vegetation is generally more abundant in the upper and more sensitive reaches, where streams are smaller and more likely to experience significant temperature changes if the vegetation is removed.

Riparian vegetation affects stream temperature in three ways (Rutherford *et al.* 1997). It adsorbs some of the incoming radiation (mostly on cloudless days), it emits long-wave radiation (mostly on cloudless nights), some of which reaches the stream, and it creates a microclimate (e.g. air temperature, humidity and wind speed), which in turn affects evaporation, conduction, ground temperature and water temperature (Rutherford *et al.* 1997). Using an estimated maximum thermal tolerance temperature based on local studies (Quinn *et al.* 1994, Quinn & Hickey 1990, Richardson *et al.* 1994), Rutherford *et al.* (1997) developed a stream water temperature model (STREAMLINE) for New Zealand headwater streams aimed at predicting the level of shade required to maintain daily maximum water temperature close below the estimated thermal tolerance level. Their study indicated that moderate shade levels of 70% cover may be sufficient to restore the temperature of headwater pasture streams to 20°C, an estimate of the thermal tolerance for sensitive macroinvertebrates (Rutherford *et al.* 1997).

4.2.4 Inter-basin water transfers

The transfer of water into cool headwater streams from warmer lower reaches may have effects on riverine biota. For example, the transfer of water from Theewaterskloof Reservoir, in the

Riviersonderend catchment of the south-western Cape, South Africa, into the cooler upper Berg River altered the biotic communities of the receiving stream (Snaddon & Davies 1998).

4.2.5 Global warming

The major anthropogenic contributions to global warming include the combustion of fossil fuels and biomass, nuclear fission, the burning of forests, and human and animal wastes (NWQMS 2000). In a study aimed at simulating the effects of global warming on stream invertebrates, Hogg *et al.* (1995) found that increased temperatures resulted in significantly lower densities of macroinvertebrates and altered growth of two species.

4.3 EFFECTS OF TEMPERATURE CHANGES ON WATER QUALITY

4.3.1 Dissolved oxygen

Higher temperatures reduce the solubility of dissolved oxygen in water, decreasing its concentration and thus its availability to aquatic organisms. If the organic loading is high, oxygen depletion is further accelerated by greater microbial activity at the higher temperature.

4.3.2 Chemical toxicity

The toxicity of many substances (e.g. cyanide, zinc, phenol, xylene), and the vulnerability of organisms to these toxins, is intensified as temperature increases (Duffus 1980).

4.3.3 Sewage fungus

Higher temperatures favour the growth of sewage fungus and the rate of putrification of sludge deposits. This reduces the environmental quality of the water as well as affecting its suitability as drinking water and aesthetic values for recreation.

4.4 EFFECTS OF TEMPERATURE CHANGES ON AQUATIC BIOTA

All organisms have a range of temperatures at which optimal growth, reproduction and general fitness occur. Several studies have documented physiological effects on individual organisms, modifications to life cycle stages and effects on communities.

4.4.1 Physiological effects

Elevated temperatures increase metabolic rate, including respiration and thus oxygen demand, of aquatic organisms (respiration approximately doubles for a 10°C rise in temperature) (McKee & Wolf 1963). Oxygen demand is therefore increased at the same time that supply decreases.

4.4.2 Modifications to life cycle stages

Many life cycle characteristics of aquatic organisms are cued into temperature. False temperature cues caused by modified temperature regimes may affect the timing of life history stages such as reproductive periods, rates of development and emergence times, and thus interfering with normal development.

An example might be the accelerated development and early emergence of insects into an environment unsuited to their existence (lack of food, ambient air temperature too low for emerging adults to survive, etc.) *Peak emergence of adult insects from an experimental stream* with an elevated temperature (≈10°C) was advanced by one to four weeks in four chironomid (midge) groups, and one damselfly species (Nordlie & Arthur 1981). The reproductive periods of one snail and one isopod species started 2-3 months earlier in the heated channel. A water temperature increase of ≈6°C (Langford & Daffern 1975), however, did not affect the emergence of insects in the River Severn, England. Aston (1973 cited by Howells 1983) noted an increase in oligochaete (*Limnodrilus hoffmeisteri, Tubifex tubifex*) numbers downstream of power station discharges, and when investigated experimentally, egg production was shown to increase as temperature increased, until a maximum of 25°C had been reached.

Arthur *et al.* (1982) sampled the macroinvertebrate communities established in two outdoor experimental stream channels: one a control, maintained at ambient temperature, and one maintained at ≈10°C higher than the control. They noted that peak macroinvertebrate density in the heated channel riffles occurred 3 to 4 weeks before that in the ambient channel. There was a seasonal alteration in peak faunal densities: greatest in winter and early spring in the heated channel, and greatest in summer in the control channel. Wrenn *et al.* (1979 cited by Nordlie & Arthur 1981) found an increased potential for additional generations of *Caenis* mayflies in channels with water temperatures elevated 4-6°C above ambient.

4.4.3 Effects on communities

Changes in water temperature may lead to changes in the abundance, diversity and composition of aquatic communities. As stenothermal species (organisms adapted to a very narrow range of temperatures) disappear from heated waters, heat-tolerant species increase in number and replace the original species in the ecosystem (Reid & Wood 1976). Studies in Texas (Wellborn & Robinson 1996) and Pennsylvania (Coutant 1962 cited by Mann 1965) showed a reduction in macroinvertebrate abundance and diversity following an average increase of 7.2°C. Effects were most pronounced in summer when temperatures of 40-42°C eliminated macroinvertebrates (Wellborn & Robinson 1996). Summer mortality in Pennsylvania occurred at 32-35°C and was confined to the side of the river where the heated effluent was concentrated. Wellborn & Robinson (1996) also noted a large bloom of blue-green algae during summer. Mann (1965) determined the changes that occurred below the Earley power station on the river Thames following a temperature increase of 10-12°C. He noted a change in the abundance of certain groups: flatworms (Platyhelminthes: Turbellaria), leeches (Annelida: Hirudinea), shrimps (Crustacea) and snails

(Mollusca: Gastropoda) decreased in number, whilst worms (Oligochaeta), mussels (Mollusca: Bivalvia) and encrusting organisms increased.

Howells (1983) reviewed various studies of temperature effects on aquatic organisms, but decided that they were not conclusive. As with many perturbations, it is the extent and characteristics of the particular perturbation that determine its effect.

4.5 ASSESSMENT OF TEMPERATURE EFFECTS ON AQUATIC BIOTA

The effects of temperature changes may be assessed in terms of an organism's lethal limit s (minimum and maximum), sublethal effects or behavioural avoidance preferences.

4.5.1 Lethal effects

Lethal effects are normally determined by acute (short-term) experimental exposure to a range of temperatures in order to measure LC_{50}'s. LC_{50}'s enable thermal tolerance levels to be established, and these may in turn provide input into predictive models such as STREAMLINE (Rutherford *et al.* 1997). Quinn *et al.* (1994) found that the lethal temperature (96-h LC_{50}) for a selection of New Zealand macroinvertebrates varied from 22.6 ± 0.9 (*Deleatidium* spp., the most sensitive species tested) to 32.5 ± 2.5 (*Potamopyrgus antipodarum*, *Pycnocentrodes aureola*, and *Hydora* spp., the three least sensitive species tested). When diurnally varying temperatures were used instead of constant temperatures, LT_{50} was 10% lower than ${}_{const}LT_{50}$ (Cox & Rutherford 2000a). Based on a combination of laboratory studies and field observations, thermal tolerance levels may be established. These are likely to be region- and reach-specific. Tolerances levels of indigenous New Zealand fish suggest that they have slightly higher thermal tolerances than the macroinvertebrates have (Richardson *et al.* 1994). Mortality in fish from acute exposure to elevated temperatures is basically the result of metabolic malfunctions (including fluid-electrolyte imbalance, alterations in gaseous exchange and osmoregulation, hypoxia of the central nervous system and inactivation of enzyme systems) (Cherry & Cairns 1982).

4.5.2 Sublethal effects

Sublethal effects include effects on growth, reproduction, survival of fry and egg hatchability, as well as sublethal physiological and community/population effects. Brungs (1971a), using 12-month exposure experiments of *Pimephales promelas* (fathead minnow) to temperatures $\approx 10^{0}$C higher than ambient, concluded that effects on reproduction (number of eggs per female, number of eggs per spawning, number of spawnings per female) were more sensitive than were growth, survival and egg hatchability.

4.5.3 Behavioural preferences and avoidance

Cherry & Cairns (1982) reviewed methods for assessing temperature changes with respect to behaviour in fish. Essentially, this technique involves the establishment of a temperature gradient that is then used to evaluate the temperatures preferred or avoided by various organisms. This method facilitates the determination of the effects of gradual temperature changes on aquatic organisms. For spawning and hatching of fish eggs, much lower temperatures are required: for instance, trout eggs do not hatch at temperatures over 14.4°C (Southgate 1951 cited by McKee & Wolf 1963). TL_M values are important to know, but in order to maintain a system in as natural a state as possible, it is necessary to know the optimal or preferred temperatures of the aquatic organisms. Gradual changes in temperature can result in the subsequent gradual alteration of species composition (particularly of fish) if their optimal ranges are exceeded, either negatively or positively.

4.6 CURRENT STANDARDS IN SOUTH AFRICA

It is clear that a change in temperature may have an effect on aquatic organisms, but that the extent and timing of the change will dictate the seriousness of the effect. Slight temperature changes, if maintained for a period of time, could lead to alteration of community composition, as a result of the differential optimal temperatures of the respective organisms. Acute, radical temperature shifts can lead to mortality of organisms by exposing them to temperatures outside their lethal limits.

Establishing suitable temperature criteria is difficult in view of the differences in temperature tolerances amongst the biota and, because of spatial differences in temperature, no single temperature would be suitable for all seasons, all parts of a country, all river zones or all species. Predicting the effects of time-varying temperatures on stream invertebrate mortality has recently been investigated for two invertebrate species in New Zealand (Cox & Rutherford 2000b).

In South Africa, the norms for assessing the effects of water temperature on aquatic ecosystems include the measurement of the acute and chronic physiological effects on aquatic organisms, and the effects of changes from "natural" site-specific temperature regimes which result in changes to ecosystem structure and functioning (DWAF 1996). Water quality guidelines for aquatic ecosystems in South Africa, therefore, specify a target water quality range (TWQR) whereby water temperature should not be allowed to vary from the background daily average water temperature considered to be normal for that specific site and time of day, by > 2°C, or by > 10%, whichever estimate is the more conservative (DWAF 1996, but also see Dallas *et al.* 1998).

CHAPTER 5: TURBIDITY AND SUSPENDED SOLIDS

SUMMARY

The immediate visual effect of a change in turbidity is a change in water clarity. An increase in turbidity or suspended solids affects light penetration, which may have far-reaching consequences for aquatic biotas. The natural seasonal variations in rivers often include changes in turbidity, the extent of which is governed by the basic hydrology and geomorphology of the particular region. Erosion of land surfaces in catchment areas by wind and rain is a continuous and historically natural process. Land-use practices such as overgrazing, non-contour ploughing and removal of riparian vegetation accelerate this erosion, however, and result in increased quantities of suspended solids in associated rivers. Increases in turbidity can, and often do, result from other anthropogenic processes, such as release of domestic sewage, industrial discharge (including mining, dredging, pulp and paper manufacturing) and physical perturbations such as road and bridge construction, dam construction, road use and reservoir management. If turbidity increases resulting from human inputs are as infrequent as natural flooding is, the stream community may tolerate them. Continuous high-level inputs, on the other hand, may have very serious consequences for the riverine biota. As light penetration is reduced, primary production decreases and food availability to organisms higher in the food chain is diminished. Suspensoids that settle out may smother and abrade riverine plants and animals. Community composition may change, depending on which organisms are best able to cope with this alteration in habitat. Predator-prey interactions are affected by the impairment of visually-hunting predators. Nutrients, trace metals, biocides and other toxins adsorb to suspended solids and are transported in this form. Few studies on turbidity effects have been conducted in South Africa, primarily because turbid rivers are fairly common in this country and are thus not considered to be problematic.

5.1 INTRODUCTION

Turbidity is the water quality characteristic most obvious to the casual observer. Its immediate visual effect is to decrease the clarity of water. This factor, together with water colour, leads to impeded light penetration, an effect that may have far-reaching ecological consequences. The major components and properties affecting light penetration of natural waters, modified from Gippel (1989), are tabulated in Table 5.1.

The American Public Health Association (1989) defines turbidity as an expression of the optical property that causes light to be scattered and absorbed rather than transmitted in straight lines through the sample. Suspended matter such as clay, silt, finely divided organic and inorganic matter, plankton and other microscopic organisms causes the scattering of light, whilst the absorption of light is caused by soluble coloured organic compounds.

Table 5.1 The major components and properties affecting light penetration in natural waters (modified from Gippel 1989).

Major components **of stream loads**	**Examples**	*Properties affecting light* **penetration**
Dissolved organic matter	Fulvic acid, humic acid, ligno-sulphonic acid, tannic acid	Some organic acids give water a yellow-brown colouration: affect colour of water
Dissolved inorganic matter	Ionic forms of minerals	Concentrations are usually too low to affect light penetration: no effect
Suspended organic matter	Pollen, micro-organisms, seeds and other fine to coarse particles	Variations in colour, shape, size, fluorescence and refractive index: affect turbidity of water
Suspended inorganic matter	Products of weathering (e.g. kaolinite, quartz, illite, smectite, mica) and speciation (eg. hydroxides)	Variations in colour, shape, size, fluorescence and refractive index: affect turbidity of water

Kirk (1985), however, defines turbidity as the extent to which the water appears to interfere with the straight-line *transmission* of light, i.e. the effect of the presence of suspended particles (suspensoids); thus the dissolved component (responsible for water colour and hence absorption of light) is not considered. Operationally, it is convenient to define suspended solids as that fraction in the water that is removed by 0.22 µm- (or perhaps 0.45 µm-) pore-size filter. By this definition, plankton would be considered as a fraction of suspended solids. It is the concentration of inanimate particles (the "tripton" fraction of limnologists), predominantly inorganic in nature and derived from soils in the catchment, that dominates in African and Australian waters and thus distinguishes them from turbid waters in the northern hemisphere, where turbidity is usually caused by organic suspended solids (Kirk 1985). For this reason, "turbidity" in the southern hemisphere is generally considered to be equivalent to some measure of the concentration of suspended solids. Suspended solids that are either washed in during rainfall events or are brought into suspension from the bottom sediments of rivers during spates seasonally inundate turbid rivers. As flow decreases, so these larger suspended solids settle out. The rate of settlement is dependent on particle size and the hydrodynamics of the water body. The true suspensoids, however, remain in suspension even in low or zero flow conditions (i.e. they are the $<$ 0.45 µm size fraction).

5.2 MEASURING LIGHT PENETRATION AND TURBIDITY

Light penetration is normally measured either *in situ* by visual observation or using a light probe, or *in vitro* using a spectrophotometer or turbidimeter. An old, but still very much utilized and valuable method of measuring light penetration (i.e. transparency), is the use of the Secchi disc, a simple disc 100mm in diameter, with alternating quadrats painted black and white. The disc is lowered into the water body and the depth at which it disappears is noted. This is approximately the depth to which 5 percent of the sunlight penetrates.

The more common of the other two methods is the nephelometric method, which is based on a comparison of the intensity of light scattered by the sample under defined conditions, with the intensity of light scattered by a standard reference suspension under the same conditions (American Public Health Association 1989). The higher the intensity of scattered light, the higher the turbidity. Originally the units of turbidity were expressed as parts per million of suspended solids, but because of variation amongst different kinds of solid particles this method has been replaced by one that relates to the optical property of the medium. Turbidity values are now expressed in Nephelometric Turbidity Units (NTUs). A colorimeter that has been calibrated using formazine may be used most successfully as a turbidimeter in pollution studies. Light penetration through a water sample is measured thereby permitting the calculation of the NTU value. Phytoplankton biologists are interested in the amount of light reaching individual phytoplankton cells, and hence in the effects of suspended solids on the underwater light regime. For this reason, a number of probes are now available for determining the relationship between, for instance, incident and refracted light within the water column. The values obtained by these probes are directly related to the turbidity of the water and are indeed sometimes used as a measure of turbidity. The exact relationship between these values and NTUs are poorly known, however. For this reason, and because NTU values are already widely available for many rivers, it is recommended that direct measurement of turbidity as NTUs is continued for pollution studies.

Suspended solids may be measured by mass, as well as by methods that rely on the optical properties they confer on water (i.e. the methods mentioned above). The total suspended solids (TSS) concentration of a water sample is measured gravimetrically and expressed as mg l^{-1}. A measured volume of water is filtered through pre-weighed, pre-dried filter paper (0.45 μm pore size), which is dried at 60°C for 48 hours and then re-weighed.

5.2.1 Relationship between turbidity and suspended solids

The optical changes resulting from the addition of particles to water will vary depending on the size and size distribution of those particles; thus measures of turbidity and TSS are not necessarily strongly correlated. From examination of studies on man-made lakes in which turbidity and suspended solids have been measured (Walmsley & Butty 1980), it appears that in at least some parts of South Africa, 1 mg l^{-1} is very roughly equivalent to 1 NTU. It is not known if this holds for rivers during base flow, and during periods of increased runoff such a relationship is unlikely to hold. Lloyd *et al.* (1987) defined a relationship for Alaskan streams whereby turbidity could be used to estimate the concentration of suspended solids. Many investigators caution that relationships between turbidity and suspended sediment concentration are useful only for specific drainages for which they were developed because relationships differ between drainages due to particular sediment characteristics. Lloyd *et al.* (1987) however, suggest that measurements of turbidity can be used to identify threshold levels of suspended sediment concentrations for a broad range of watersheds.

Gippel (1989) tested the relationship between turbidity and suspended solids to determine if one could use a turbidimeter for accurate monitoring of suspended solid concentrations continuously during runoff events. He noted that there was not always a linear relationship between the turbidity measurements and suspended solid concentrations, that the variance was high, and that this relationship was often site-specific since the amount and size of the suspended solids varied from site to site. He concluded, however, that by calibrating with careful consideration of site-specific characteristics, turbidimeters could provide the opportunity for greater understanding of sediment dynamics in storms. For some studies the determination of the concentration of suspended solids is extremely useful, but it is important to bear in mind that increased sediment loads are not necessarily linearly related to increased turbidity, particularly during storm events.

5.3 POTENTIAL SOURCES OF INCREASED TURBIDITY AND SUSPENDED SOLIDS

The natural seasonal variations in rivers often result in changes in turbidity (Harrison & Elsworth 1958), the extent of which is governed by the basic hydrology (e.g. flow regime, rainfall) and geomorphology (e.g. weathering, aspect of slopes) of the particular region. Studies in the Murray-Darling river system of Australia have shown the influence of these natural factors on particular sections of the rivers (Shafron *et al.* 1990). The supply of sediment from the two main sources, namely channel sources and catchment sources, has been comprehensively summarised by Wood & Armitage (1997).

Channel sources of suspensoids are strongly related to stream discharge and the stability of the channel-bed and banks. Channel sources include 1) river banks subject to erosion during periods of high discharge; 2) mid-channel and point bars subject to erosion; 3) fine bed material stored within the interstices; 4) natural backwaters where sediment accumulates during base flow conditions; 5) fine particles trapped within aquatic macrophyte stands; and 6) organic particles including phytoplankton and zooplankton. Catchment sources, in contrast, may be highly variable depending on its mode of production and transport into the stream. Catchment sources of fine sediment supplied to a stream include 1) exposed soils subject to erosion; 2) landslides and other mass failures within the catchment; 3) urban areas, 4) anthropogenic activities; 5) litter fall; and 6) atmospheric deposition, due to aeolian processes and precipitation.

Erosion of land surfaces by wind and rain is a continuous and historically natural process. Humans can increase the mobilisation of large volumes of sediment into streams and rivers by activities such as agriculture (Richards *et al.* 1993) (e.g. land use practices such as removal of riparian vegetation, overgrazing, non-contour ploughing), forestry operations (Scrivener & Brownlee 1989), physical perturbations such as road and bridge construction (Barton 1977, Chisholm & Downs 1978, Horner & Welch 1982, Ogbeibu & Victor 1989, Taylor & Roff 1986), dam construction (Chessman *et al.* 1987, Blyth 1986), road use (Smith & Kaster 1983) and reservoir management (Gray & Ward 1982a, b). Anthropogenic processes include the release of domestic sewage and industrial discharges from mining, dredging, pulp and paper manufacturing and china-clay industries. Channel sources of sediment are largely linked to channel incision, which is often caused by increased runoff in the catchment after clearing (Boulton & Brock 1999).

In South Africa, all rivers, excluding some in the Natal foothills of the Drakensberg and in the south-western Cape, become highly turbid and laden with suspended solids during the rainy season (Chutter 1969). Britton *et al.* (1992) showed that even in south-western Cape rivers, the loads of suspended solids increase during storm events. The Orange River carries an enormous sediment load eroded from the mountains and foothills of Lesotho, the rich farmlands of the eastern Orange Free State, the northern Cape and the Kalahari dune-field (Davies & Day 1998). Prior to construction of two major dams, the Gariep (formerly H.F Vervoed) and Vanderkloof (formerly P.K. Le Roux), on the Orange River, this sediment load was carried to the river mouth, but now it becomes trapped in the dams, at great economic cost.

5.4 EFFECTS OF INCREASED TURBIDITY ON AQUATIC ORGANISMS

The effects of elevated turbidity and deposition of fine sediments have been widely documented. The main effects are on primary production and biotic abundance and diversity. The extent of impact is also somewhat dependent on the type and duration of input. Specifically, if turbidity increases resulting from human input are as infrequent as natural flooding, the stream community may tolerate them (e.g. Cooper 1988). Continuous high-level inputs often associated with agriculture and surface mining activity (Wood & Armitage 1997), on the other hand, may have very serious consequences for the riverine biota and may lead to a complete change in the natural faunal assemblage.

5.4.1 Primary production

In turbid waters light penetration is reduced (Kirk 1985), leading to a decrease in the rate of photosynthesis and therefore of primary productivity within the stream. The decrease in primary production reduces food availability for aquatic organisms higher up the food chain. Turbidity levels as low at 5 NTU can decrease primary production in New Zealand streams by 3 -13% (Ryan 1991). Since primary producers such as periphyton and macrophytes form the base of the food chain, any deleterious impacts will probably also be manifested in the invertebrate and fish communities (Wood & Armitage 1997).

5.4.2 Benthic invertebrates

The magnitude of the increase in turbidity and the tolerance levels of individual organisms are determinants of the resulting effects on community structure. Increased turbidity and deposition of fine sediment may affect benthic invertebrates in four ways (Wood & Armitage 1997): 1) by altering substrate composition and changing the suitability of the substrate for some taxa; 2) by increasing drift due to sediment deposition or substrate instability; 3) by affecting respiration due to deposition of silt on respiratory structures; and 4) by affecting feeding activities by impeding filter feeding, reducing the food value of the periphyton and reducing the density of prey items. Temperature-sensitive species may also be affected since increased turbidity often results in a reduction in water temperature as more heat is reflected from the surface and less absorbed by the water.

Hellawell (1986), in a review of physical disturbances on stream communities, summarized the effects as follows:

"Certain recurrent features will have been noted with respect to the effects of deposition of solids on the benthos. These include the lowering of benthic community diversity through the disappearance or marked reduction in biomass and numbers of certain sensitive species, often those requiring 'open' eroded substrata for attachment or feeding (especially filter feeders). The replacement fauna consists of burrowing forms, typical of soft, depositing substrates, provided that the deposits are neither excessive in their rates of accumulation or completely sterile and devoid of nutriment. Almost invariably the dominant organisms are chironomid larvae, specifically Chironominae (Angradi 1999), and oligochaete worms and whenever the deposits are also organically enriched then these groups may become extremely abundant".

As suspended solids settle out they may smother or abrade benthic fauna, particularly if this "settling out" occurs in a biotope such as a stones-in-current (riffle) area, where organisms are generally not tolerant of these deposits. Harrison & Farina (1965 cited by Chutter 1969) observed egg-laying and development of three species of planorbid snails, and noted that two of these, *Biomphalaria pfeifferi* and *Bulinus globulus*, were severely and adversely affected by suspended material. Wu (1931 cited by Chutter 1969) noted that simuliid larvae and certain mayflies (*Pseudocloeon vinosum*) do not tolerate silty surfaces. If one examines the anatomy of most of these organisms, it is easy to understand why suspended and settled material would interfere with their respiratory and feeding apparatuses.

Increased levels of turbidity have been shown to increase drift (i.e. the rate at which organisms move by floating downstream), which in turn leads to a decrease in the density of benthic organisms (Ryder 1989 cited by Ryan 1991). Studies have shown that benthic invertebrate assemblages often decline in abundance and taxa diversity in response to sediment deposition (e.g. Campbell & Doeg 1989, Davies & Nelson 1994), although Chessmann *et al.* (1987) concluded that low levels of sediment deposition did not seriously affect invertebrate populations, even though a decline in some trichopterans was noted. Rosenberg & Wiens (1978) experimentally increased sediment load to the Harris River (Australia) and showed that it led to an increase in invertebrate drift. They could not correlate the actual response to the sediment level, suggesting that some threshold value may be involved.

A recent study examined the response of benthic stream invertebrates in the USA to surficially deposited fine sediment (Zweig & Rabeni 2001), specifically the response of biomonitoring metrics to sediment deposition. The authors showed that the density of invertebrates was significantly correlated with the quantity of sediments deposited, densities decreasing substantially before 30% of the substratum had been covered, as estimated both visually and by measuring embeddedness. Of significance is the absence of a relationship between measures of deposited sediments and tbiotic indices (Angradi 1999, Zweig & Rabeni 2001), prompting the development of a Deposited Sediment Biotic Index (DSBI, Zweig & Rabeni 2001), which is specifically aimed at assessing the effect of deposited sediment on benthic invertebrates.

5.4.3 Fish

Wood & Armitage (1997) summarised ways in which fish are adversely affected by elevated turbidity and suspended sediment concentrations. They are: 1) physiological effects on the fish such as impairment of gill function or reduced resistance to disease; 2) reduction in the suitability of spawning habitat and hindering of the development of eggs, larvae and juveniles; 3) modification of migration patterns; 4) reduction in food availability through decreased primary production and habitat loss; and 5) interference with hunting efficiency of fish, particularly in the case of visually hunting predators.

The effects of suspended solids on fish include impairment of gill function and reduced foraging efficiency and growth, but also include increased protection from avian and piscine predators (Bruton 1985). Fish populations have exhibited stress due to the presence of silt particles, coupled with low dissolved oxygen levels induced by the silt. Sublethal levels of suspended solids resulted in an increase in oxygen consumption by fish due to an increase in metabolic rate associated with increased activity and stress (du Preez *et al.* 1996).

Suspensoids that settle out may transform in-stream habitat such as cobble or gravel beds, by filling in the interstitial spaces. Notable is the siltation of fish spawning areas (Doeg & Koehn 1994, Petticrew 1996 cited by Ashmore *et al.* 2000), in particular the spawning areas of substrate spawners (Muncy *et al.* 1979 cited by Henley *et al.* 2000). Deposited suspensoids that decrease available spawning habitats lead to a reduction in spawning activity and increased egg and larval mortality (Ryan 1991). Turbidity affects the food-searching ability of visually-hunting predators (Bruton 1985), which may lead to changes in species assemblages (Lloyd *et al.* 1987, Scholtz *et al.* 1988).

Quinn *et al.* (1991) investigated the effect of inert suspended sediments on the river ecosystems of six shallow, West Coast streams in South Island, New Zealand. The primary source of turbidity (average increase of 7 - 150 NTU) was mining. Discharge of fine sediment typically reduced light penetration and algal abundance, leading to a 55-90% drop in invertebrate abundance below the mines. The impact of the mines increased with the increase in turbidity and appeared to be very strongly correlated with the reduction in algal content of the inorganic material on stones. Fish responses varied: some were notably negatively affected whilst others appeared to prefer turbid reaches, presumably because of the increased protection afforded them from predators.

5.4.4 Adsorption

Nutrients, trace metals, biocides and other toxins adsorb onto suspended solids and are transported in this form. This can be beneficial (adsorbed toxins may effectively be "lost" to the system) or detrimental (adsorbed nutrients may become unavailable).

5.5 CURRENT STANDARDS IN SOUTH AFRICA

Few studies on the effect of turbidity have been conducted in South Africa and, because turbid rivers are relatively common, it has generally been accepted that turbidity is not a particularly significant water quality variable in this country. A study conducted on the Lourens River in the south-western Cape, however, investigated the effect of suspended solids resulting from detrimental agricultural and land drainage practices on invertebrate assemblages during winter (Ractliffe 1991). An increase in suspended solids was accompanied by the loss or drastic reduction in invertebrate species characteristic of mountain streams and upper river zones. The loss of ephemeropteran (mayfly) nymphs was particularly noticeable. With the ever-increasing pressure on our natural water resources from road construction etc., it is clear that more such detailed and specific studies on the effects of suspended solids and turbidity are needed.

The norms for assessing the effects of TSS concentration on aquatic ecosystems include the measurement of the acute and chronic physiological effects on aquatic organisms, and the establishment of changes from "natural" site-specific TSS levels that cause changes to ecosystem structure and functioning (DWAF 1996). It is, therefore, important to know the "natural" or background turbidity levels for a particular water body at a particular time. Elevated levels of turbidity and TSS will have a greater effect in areas, for example upper mountain catchments, that have low background TSS levels (DWAF 1996). In South Africa, in a preliminary review of suspended solid concentrations, Brown (1991) recommended a maximum TSS concentration of 5 mg l^{-1} for rivers in the sensitive mountain streams and upper reaches of rivers in the south-western Cape. Water quality guidelines for aquatic ecosystems in South Africa therefore specify a target water quality range (TWQR) whereby any increase in TSS concentration (or turbidity) should be less than 10% of the background TSS concentrations at a specific site and time. This is applicable for sites where the background TSS concentration is less than 100 mg l^{-1}.

The recovery of a stream affected by high sediment deposition is dependent on the elimination of the sediment source and the ability of the stream water to flush out the deposited material (Luedtke *et al.* 1976 cited by Ryan 1991), particularly after flood spates (Hellawell 1986). Henley *et al.* (2000) emphasised the importance of riparian buffer strips and livestock fencing to reduce input of sediments into river systems. The importance of riparian zones is discussed further in chapters 13 (agriculture) and 14 (forestry).

CHAPTER 6: pH AND ALKALINITY

SUMMARY

pH is determined largely by the concentration of hydrogen ions (H^+), and alkalinity by the concentrations of hydroxyl (OH^-), bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}) ions in water. Addition of acid or alkali to a water body alters pH. Since pH is a log scale, a change of one unit means a ten-fold change in hydrogen ion concentration. Further, in very pure waters pH can change rapidly because the rate of change is determined by the buffering capacity, which in turn is usually determined by the concentration of carbonate and bicarbonate ions in the water. The pH of natural water is determined by geological and atmospheric influences. Most fresh waters are relatively well buffered and more or less neutral, with pH ranging around 6-8. pH determines the chemical species (and thus potential toxicity) of many elements in water. For instance, aluminium is mobilized following acidification. Changing the pH of water changes the concentration of both H^+ and OH^- ions, which affects the ionic and osmotic balance of aquatic organisms. Relatively small changes in pH are seldom lethal, although sublethal effects such as reduced growth rates and reduced fecundity may result from the physiological stress placed on the organism by increased energy requirements in acid or alkaline waters. Human-induced acidification of rivers is normally the result of industrial effluents, mine drainage and acid precipitation. Alkaline pollution is less common but may result from certain industrial effluents and anthropogenic eutrophication. The effects of altered pH on riverine biotas have been investigated by means of toxicity tests, artificial streams and field studies. Such studies indicate that a change in pH from that normally encountered in unpolluted streams may have severe effects on the biota but that the severity of the effects depends on the magnitude of change. Some streams are naturally far more acidic than others and their biotas are adapted to these conditions. Water quality guidelines require that the Target Water Quality Range for pH be stated in terms of the background site-specific pH regime. Guidelines are thus case- and site-specific and take diel and seasonal variation into account. pH values should not be allowed to vary from the range of the background pH by > 0.5 of a H unit, or by > 5%.

6.1 INTRODUCTION

The concentrations of hydrogen (H^+), hydroxyl (OH^-), bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}) ions are some of the most important attributes determining the composition and quality of water. Furthermore, these ions are in dynamic equilibrium in most water samples, and a change in the concentration of any one will have an effect on all of the others. In this chapter we discuss the concepts of pH, alkalinity and buffers, the interactions between them in solution, and their effects on the chemistry and biotas of aquatic ecosystems.

6.2 pH, ALKALINITY AND BUFFERING CAPACITY

6.2.1 pH

The theory behind the chemistry of pH, alkalinity and buffers is complex. The reader is referred to Golterman *et al.* (1978) for a brief analysis and to Stumm & Morgan (1970) and Schnoor & Stumm (1985) for more detailed treatments of the subject.

Pure water (i.e. water containing no solutes) ionizes very slightly to produce a small proportion of hydrogen and hydroxyl ions:

$$H_2O \leftrightharpoons H^+ + OH^-$$

In fact, the hydrogen ions combine with water molecules to form H_3O^+ (hydronium) ions but for convenience we will refer only to the hydrogen ions themselves. The concentration of both H^+ and OH^- ions is exactly 10^{-7} mol l^{-1} at 24°C. Because these concentrations are very low, it is more convenient to convert them to a log scale. pH is defined as the negative $\log_{10}$ of the hydrogen ion activity:

$$pH = -\log_{10}[H^+]$$

(The term "activity" refers to the "apparent" concentration of ions in non-ideal solutions - i.e. solutions where ions interact with each other, as they do even in fresh water. For practical purposes, though, activity and concentration are virtually identical in fresh waters and so the term "concentration" will be used from here on.)

Since $[H^+]$ is 10^{-7} mol l^{-1} at 24°C, the pH of pure water at 24°C is 7.0. $[OH^-]$ is also 10^{-7} mol l^{-1}, so that there are equal numbers of OH^- and H^+ ions (i.e. the pOH is also 7.0) and the water is electrochemically neutral. As $[H^+]$ increases, so pH decreases and the solution becomes more acid; for instance, at a pH of 4, the solution contains 10^{-4} mol l^{-1} of H^+ ions (and 10^{-10} mol l^{-1} of OH^- ions). As $[H^+]$ decreases, pH increases and the solution becomes more alkaline: at a pH of 10, say, the solution contains 10^{-10} mol l^{-1} of H^+ ions (and 10^{-4} mol l^{-1} of OH^- ions).

It should be noted that the degree of dissociation of water into H^+ and OH^- ions depends on the temperature. Thus, although the pH of pure water is 7.0 at 24°C, at 0°C it is 7.47 and at 30°C it is 6.92. This effect is less marked in natural waters because they are buffered to some extent. The pH of fresh water decreases by about 0.1 unit for a temperature increase of 20°C (Golterman *et al.* 1978). In consequence, it is not necessary to correct for temperature when measuring pH in most cases, although errors can be significant where very accurate measurements of pH are required.

6.2.2 Alkalinity

Carbon dioxide dissolves in water to form carbonic acid (H_2CO_3), which, depending on pH, dissociates to form carbonate, bicarbonate and hydrogen ions:

$$CO_2 + H_2O \leftrightharpoons H_2CO_3 \leftrightharpoons HCO_3^- + H^+ \leftrightharpoons CO_3^{2-} + 2H^+$$

Figure 6.1 illustrates the relationship between pH and the percentage of free carbon dioxide, bicarbonate and carbonate ions in water. Note that below a pH of 4, all of the CO_2 is in the form of CO_2; above this pH the proportion of bicarbonate rises to a peak at a pH of 8.3, when carbonate ions begin to appear. In practical terms, most CO_2 in natural waters is in the form of HCO_3^-, while CO_3^{2-} is present in significant quantities only at pH values approaching10. The term *alkalinity* refers to the sum of the anions (OH^-, CO_3^{2-}, HCO_3) of weak acids, plus hydroxyl ions and bicarbonate, in a sample of water (Mackereth *et al.* 1978) and is thus a measure of the amount of acid required to bring a sample of water to the inflection points shown in Figure 6.1. It is thus usually an estimate largely of the concentration of bicarbonate and carbonate ions, although other ions may contribute a small amount in some waters. Alkalinity is usually determined either potentiometrically or by titration and is often quoted, in units of carbonate, bicarbonate or total alkalinity, as "mg l^{-1} calcium carbonate", although these days it is more acceptable to use mmol l^{-1} or meq l^{-1} HCO_3^- and/or CO_3^{2-}. The term *acid neutralizing capacity* is commonly used in studies on the effects of acid precipitation on aquatic ecosystems. It appears to be analytically and quantitatively the same as alkalinity. For a useful discussion of alkalinity and its measurement, see Neal (2001).

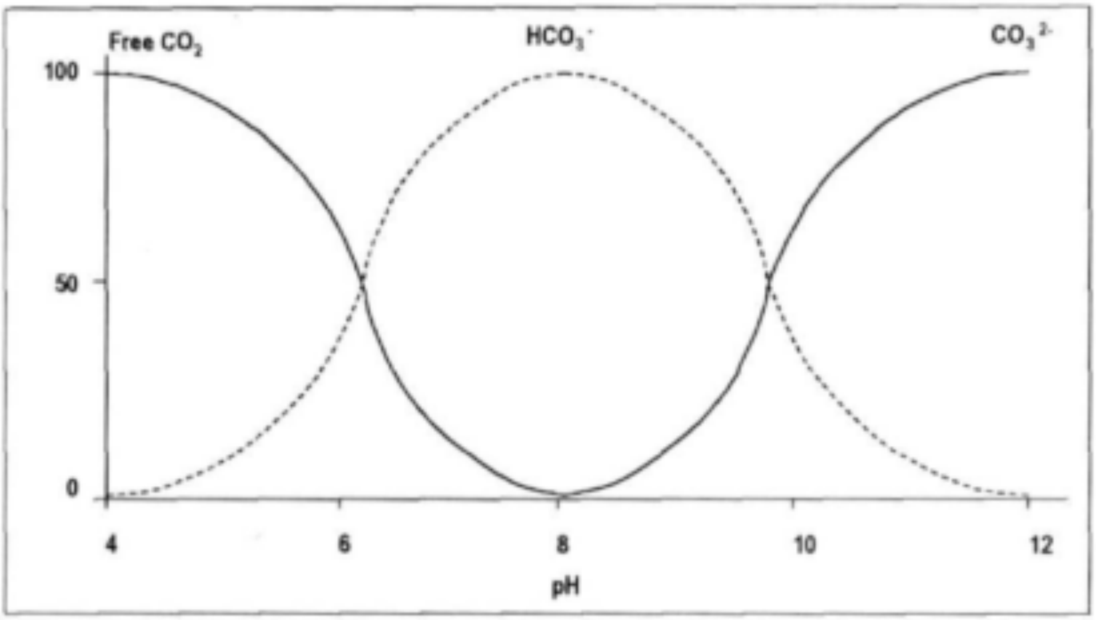


Figure 6.1 The relationship between pH and the percentage of free carbon dioxide, bicarbonate and carbonate ions in water (from Golterman *et al.* 1978).

6.2.3 Buffering capacity

If a given quantity of an acid or an alkali is added to pure water, the pH changes rapidly. If the water is not pure, the rate of change is generally less rapid. This is because certain salts act as *buffers*. Essentially a weak acid (or a weak base) plus its salts in water form a buffering system because the ionic equilibrium alters between the acid (or base) and its salts as pH changes, in a similar manner to that shown in Figure 6.1. The most important buffering system in fresh waters is the carbonate-bicarbonate one. Thus in a sense, alkalinity can be thought of as a measure also of buffering capacity. In naturally acid waters, complex polyphenolic organics and their salts may form the major buffering system, while aluminium and its salts become the effective buffering agents in waters made very acid by in acid precipitation (see section 6.5.1.10).

6.3 NATURAL CONDITIONS

The relative proportions of the major ions in, and in consequence the pH of, natural waters, are determined by geological and atmospheric influences (see Chapter 2). Many fresh waters, including most in South Africa, are relatively well buffered and more or less neutral, with pH ranging around 6-8. Very dilute NaCl-dominated waters (see Chapter 2) are poorly buffered because they contain virtually no bicarbonate or carbonate ions. If they drain catchments containing certain types of vegetation (e.g. fynbos, some forest types), the pH may naturally drop as low as 3.9 owing to the influence of organic acids (e.g. humic and fulvic acids and other polyphenol-rich compounds) leaching from the vegetation (Gardiner 1988, Silberbauer & King 1991a, Britton 1991, Gale 1992). These waters are usually also very dark in colour. In South Africa such conditions are found particularly in parts of the south-western and southern Cape coastal belt and the swamp forests of Natal. On the other hand, highly alkaline lakes may result from unusual geological features (e.g. soda lakes such as Etosha Pan in Namibia) but river waters, certainly in South Africa, are seldom naturally very alkaline (pH >9) (Day 1993).

Extreme rates of photosynthesis, whether natural or as a result of eutrophication (see Chapter 10), commonly cause very high pH values in standing waters. High rates of consumption of CO_2 during photosynthesis may drive the CO_2/bicarbonate/carbonate equilibrium far towards carbonate and hence to extremely high pH values (>10). This process can occur only in sunlight. At night, the major biotic processes are respiration and decomposition, which release CO_2 and result in a decrease in pH. Thus the pH of very eutrophic systems may exhibit wild diurnal fluctuations from <6 to >10. Rivers seldom show these large fluctuations in pH because they rarely exhibit the extremes of eutrophication encountered in standing waters.

It should be pointed out that the spot measurement of pH is almost valueless in systems with significant biological activity. In order for pH measurements to be of any use in the assessment of water quality in such systems, they must at least be taken under comparable conditions of ambient light and temperature; measurements should be taken at night or over a 24-hour period. Because of the problems associated with the interpretation of pH measurements, many biologists are sceptical about the use of pH in the assessment of water quality in general. In fact, though, a continuously low pH value is probably the best single measure of the integrity of naturally acid, blackwater systems and is of great value in assessing the quality of their waters. Thus although great care must be taken in the interpretation of any pH measurements, they can be of great value in water quality monitoring.

6.3.1 Natural acidification

Although acidification of soils and natural waters by "acid rain" is presently of particular concern (see section 6.5.1.), it should be pointed out that acidification can also be a natural process, resulting in widely scattered regions of acid soils and low-pH waters around the globe. Sutcliffe (1983), for instance, points out that soil acidity probably first developed in the English Lake District some time between five and seven thousand years ago. Although the processes involved are complex, in essence, acidification is likely to occur wherever soils lack base cations, particularly Ca^{2+} and Mg^{2+} (Cresser & Edwards 1988). The process is exacerbated when low-nutrient soils encourage the growth of vegetation types that produce large amounts of secondary plant compounds, particularly polyphenolics, that decompose to form humic and other weak organic acids. A typical example is the fynbos region of the south-western Cape (e.g. Raubenheimer & Day 1991).

6.4 THE EFFECTS OF CHANGE IN pH

6.4.1 Effects on water chemistry

The pH of a sample of water determines the particular chemical species in which many elements are found in that sample. This is perhaps most obviously seen in relation to proteins, and numerous other organic molecules, which are amphoteric, being able to exist as bases or as acids, depending on pH. It also determines the chemical species, and thus availability and toxicity, of metals in water. For instance, metals such as aluminium occur as unavailable hydrated hydroxides in alkaline conditions but as pH falls below neutral, they form available and highly toxic ions such as the aqua-Al^{3+} ion. The metals most likely to have environmental effects as a result of lowered pH are silver, aluminium, cadmium, cobalt, copper, mercury, manganese, nickel, lead and zinc (Campbell & Tessier 1987, who provide a useful review of the topic).

Non-metallic ions can be similarly affected by changes in pH. For instance, ammonium ions (NH_4^+) are not toxic; indeed, ammonium is the form in which nitrogen is assimilated by most plants. At a pH above about 8, however, they are gradually converted to the highly toxic un-ionized ammonia (NH_3^+)(see Table 10.2). Cyanide, too, become more toxic in acidic conditions. In consequence, pollutants that change the pH of water can profoundly increase the toxicity of otherwise innocuous substances. Lowering pH can also decrease the solubility of certain elements such as selenium. Selenium is an essential element (Mushak 1985) and concern has been expressed that human populations from areas particularly polluted by acid rain may begin to suffer from Se deficiency (Svensson *et al.* 1987).

Since the adsorptive properties of large molecules (such as polyphenolics) and of particulate material in water depend on their surface charges, altering the pH can also alter the degree to which nutrients such as PO_4^{3-}, trace metals and biocides adhere to these materials. Such an effect is of particular significance where lowered pH can lead to the release of toxic substances from sediments.

6.4.2 Physiological effects of pH on aquatic organisms

Essentially, changing the pH of water results in a change in the concentration of both H^+ and OH^- ions. This in turn affects the rate and type of ion exchange across body surfaces, particularly gills. Thus the direct effect of a change in pH is an alteration in the water, ionic and osmotic balance of individual whole organisms (e.g. Wood & Rogano 1986). Freshwater organisms generally have well developed abilities to maintain ionic and osmotic balance within rather narrow limits. For this reason, it is thought that the direct effects of alterations in pH are not normally the most important in determining the clearly detrimental consequences of relatively small changes in pH on freshwater organisms (Sutcliffe 1983). There is no doubt that the need to increase the rate of osmotic and ionic regulation places physiological stress on organisms by increasing energy requirements. This can in turn have sublethal effects such as slow growth and reduced fecundity (e.g. Berrill *et al.* 1991).

It is possible that lowering the pH can be directly detrimental to aquatic organisms, most likely in very dilute waters, where organisms are already ionically stressed. Under these conditions, hydrogen ions may fatally compete with larger cations such as Na^+ (Sutcliffe 1983).

The indirect effects of decreased pH (i.e. elevation in $[H^+]$) can be far more important. It is generally considered at present that the major effect of acidification is the mobilization (and hence availability) of toxic substances, particularly aluminium (see section 6.5.1.1.).

6.5 HUMAN INFLUENCES ON THE pH AND ALKALINITY OF INLAND WATERS

Human-induced acidification of aquatic ecosystems is normally the result of one of three different types of pollution. Firstly, low-pH point-source effluents are produced by, for instance, the chemical, pulp and paper, and tanning/leather industries (see Chapter 18). Secondly, mine drainage water (see Chapter 19) is nearly always exceedingly acid, the pH of receiving streams sometimes dropping to <2. Thirdly, air pollution can result in acid precipitation ("acid rain").

Alkaline pollution of rivers is less common than acid pollution. Increases in pH can result from certain alkaline effluents from industries such as food canning and textile production, as well as from anthropogenic eutrophication when excessive primary production leads to depletion of CO_2 from water in the presence of sunlight.

6.5.1 Acid precipitation ("acid rain")

Rain naturally has a pH of about 5.6 because of the CO_2 dissolved in it. Some decades ago, it became clear that the pH of rain falling in the more industrialized areas of the world, particularly northern North America and northern Europe, was considerably less than 5.6 (e.g. Matschullat *et al.* 1992; Wigington *et al.* 1992). This decrease is now known to be the result of atmospheric pollution by sulphur dioxide (SO_2), largely from burning of coal, and various nitrogen oxides (NO_xes), largely from the exhausts of combustion engines. Both SO_2 and NO_xes, when dissolved in water, ultimately form strong mineral acids (sulphuric and nitric acids respectively). When acid rain falls on a catchment, the strong acids leach calcium and magnesium from the soil and interfere with nutrient availability. In water, they reduce pH and alkalinity. The effects of acid rain were first noticed in the lakes of Scandinavia, where fish populations declined and sometimes died out completely as a result of the decrease in pH of the water. Soon after that, acid rain was implicated in the die-back of hardwood forests over several parts of Europe. Although the biotic effects are not yet obvious in South Africa (Bohm 1985), acid precipitation has been recorded in several regions, most noticeably on the Eastern Transvaal highveld (Tyson *et al.* 1988; Skoroszewski, 1989; Galpin & Turner 1999). Galpin and Turner's data show increases in $[H^+]$ at all of their sites (in the north-east of the country) from the mid-1980s to the mid-1990s but they consider the decrease in pH to be attributable to an increase in sulphate concentration only for a site at Warden in the Free State; the increase in $[H^+]$ is ascribed to loss of biomass (including base cations) as a result of veld burning.

6.5.1.1 Aluminium and acid rain

One of the most fascinating effects of acidification on aquatic ecosystems is the concomitant effect of aluminium. The following brief account is taken mainly from Driscoll & Schecher (1990), with some details provided by Litaor (1987).

It is known both that Al constitutes about 8% of the Earth's outer crust and that it can be extremely toxic. The interesting fact is that, despite its ubiquity, Al is not normally available in large quantities or in toxic form in the aquatic environment. (Indeed, Al cooking utensils and cooking foil are found in most western kitchens.) This is because Al is not toxic in its metal form, nor is it available in the lithosphere, where it is largely in the form of alumino-silicate minerals that are too insoluble to participate readily in biogeochemical reactions at circum-neutral pH ranges. The speciation (and hence solubility and availability) of Al is strongly pH-dependent, however. At low pH values, Al is largely in the aquo form (Al^{3+}), which is both soluble (and therefore available) and very toxic. As pH increases, hydrolysis of Al results in a series of increasingly insoluble OH^- complexes ($Al(OH)^{2+}$ and $Al(OH)_2^+$). The solubility of Al is in fact lowest at a pH around 6.5 because above this pH, the more soluble $Al(OH)_4^-$ is formed. This is the species that predominates in circum-neutral waters. In all, it has been estimated that the mean value for total soluble Al in fresh waters is about 9μmol l^{-1} (Bowen 1966).

F^- and SO_4^{2-} ions both form complexes with Al under acidic conditions, as does Si and a variety of organic solutes, particularly those known as the "humic substances". (This is why Al slurries are used in water purification works to precipitate the humic and other polyphenolic organics that impart a dark colour to some naturally acidic waters.) Depending largely on the ions present, and pH, Al may thus be bound in complex form and hence unavailable. In other words, humic substances (and other substances that complex with Al) may reduce the toxicity of Al at low pH (e.g. Backes & Tipping 1987).

A further consequence of the combined effects of low pH and increased solubility of Al is of considerable significance, but is not generally appreciated. Under acidic conditions, the carbonate/bicarbonate buffering system is no longer operational (all inorganic carbon being in the form of CO_2), and yet the buffering capacity of acidified waters *increases* at pH values <5. This is because the various species of Al and their salts and complexes act as buffers and in fact form the buffering system in these very acid waters. Driscoll & Schecher (1990) even suggest that the lower limit of pH observed in acidic waters may be regulated by the dissolution of Al, which forms an important component of acidity in these waters. This means that sufficient base has to be added to an acidified water to neutralize the effects of Al before any increase in pH can occur. Further, when water is oversaturated with Al, as it often is at these pH values, Al oxyhydroxides form microcrystalline particles with vast surface areas. These particles adsorb a variety of organic and inorganic solutes, the most important of which from an environmental point of view may be PO_4^{3-}, which becomes unavailable as a nutrient.

In summary, the chemical effects of Al in acidified waters include the formation of toxic species of Al, the buffering at low pH by Al and its salts and complexes, the complexation (and sometimes precipitation) of polyphenolics, and hence a change in water colour, and the adsorption (and hence removal as available) of PO_4^{3-}.

It should be pointed out that atmospheric pollution is not the only source of human-induced acidification of catchments. Logging, or any other activity that removes large amounts of vegetation, removes base cations (i.e. Ca^{2+} and Mg^{2+}) that have been taken up by the vegetation. If the soils are thin or poorly buffered and poor in these ions, then their removal can be sufficient to cause acidification. The consequences are similar to those of acid rain. It has even been suggested that the death of forests in parts of central Europe may be at least partly attributable to logging over many centuries. Cresser and Edwards (1988) review the topic.

6.5.1.2 Aluminium toxicity

A huge literature is developing on the subject of aluminium toxicity (see section 12.2.1). Some of the more useful references include Burrows (1977), Baker & Schofield (1982), Hall *et al.* (1987) and reviews by Herrmann (1987) and Driscoll & Schecher (1990).

The exact mechanism whereby Al exerts its toxic effects is not yet clear. It is known that decreased pH leads to disturbances in acid-base and water balance in fish. Although most fish have physiological mechanisms that rectify such disturbances, these mechanisms require the expenditure of additional energy. As a result, growth and reproduction may be retarded at sublethal pH values (Wendelaar *et al.* 1986). It is also known that organisms such as fish and benthic insects are particularly sensitive to increased levels (>4-8μmol l^{-1}) of available Al, which has been implicated as the primary factor controlling survival of brook trout (*Salvelinus fontinalis*) in the acid-stressed Adirondack lakes in New York state, USA (Schofield & Tjornar 1980). The mechanism of toxicity seems to be related to interference with ionic and osmotic balance (e.g. Muniz & Leivestad 1980) and with respiratory problems resulting from coagulation of mucus on the gills (e.g. Schofield & Tjornar 1980). It seems that the negative charge on the mucus on the gill surfaces of fish also causes Al to accumulate there, where it interferes with the excretion of NH_3 by changing membrane pH.

Al is known to be toxic to various invertebrates and to plants, although the exact mechanisms are not understood in these cases either. It is known that Al somehow interferes with Ca^{2+} metabolism, leading Haug (1984) to suggest that Al binding alters the function of the calcium-regulating protein, calmodulin, in both plants and animals. On the other hand, preliminary experimental work by Havens (1990) on acid-sensitive (*Daphnia galeata* and *D. retrocurva*) and acid-tolerant (*Bosmina longirostris*) species of cladocerans suggests that Al binds to the maxillary glands, the major sites of ion exchange in cladocerans. Havens suggests that the result (as is the case with elevated $[H^+]$: Havas & Likens 1985) is interference with Na^+ homeostasis, which leads to neuromuscular dysfunction.

6.6 EFFECTS OF ALTERED pH ON AQUATIC BIOTAS

6.6.1 Toxic effects

Numerous *in vitro* toxicity tests have been performed in order to ascertain the effects of changes in pH on a variety of aquatic organisms. Essentially, it has been shown that some species are more sensitive than other, closely related species (e.g. Havens 1990 on cladocerans); and that the effect of pH is often really the effect of changing speciation of toxins such as trace metals (e.g. Driscoll & Schecher 1990). A variety of solutes act as Al antagonists, reducing its toxic effects (e.g. calcium: Brown 1983; and humic substances: Backes & Tipping 1987). Presently more work is being done on community effects using mesocosms in the wild, and a number of these studies are reported on below. A few representative *in vitro* studies are briefly mentioned here.

Some of the earliest studies (Bell & Nebeker 1969; Bell 1971) examined the effects on Minnesotan benthic invertebrates of reduced pH resulting from acid mine drainage. 96-hour TL_{50} values ranged from 4.65 for mayfly nymphs to 1.5 for caddisfly larvae, while 30-day TL_{50} values ranged from pH 2.45 for a caddisfly larva (*Brachycentrus americanus*) to 5.38 for the nymphs of the mayfly *Ephemerella subvaria*.

Crustaceans seem to be less sensitive: DiStefano *et al.* (1991) found the crayfish *Cambarus bartonii* to be remarkably tolerant to acute low pH: 96-hour LC_{50} values ranged from 2.43 for adults to 2.85 for early juveniles.

Numerous papers have been written about the toxic effects of pH on fish, some of the most useful being those of Driscoll *et al.* (1980), Baker & Schofield (1982), Ingersoll *et al.* (1985), Wendelaar *et al.* (1986), Henriksen *et al.* (1987) and Hall (1987). We confine the present review largely to effects on whole communities and ecosystems and do not discuss *in vitro* toxicity studies any further.

6.6.2 Experimental and mesocosm studies

Changes in pH have wide-ranging effects on water chemistry, and therefore on aquatic ecosystems, but are difficult to quantify because pH can vary cyclically over periods as short as 24 hours. Further, the generation of data on community structure, particularly of stream macroinvertebrates, is time-consuming and the interpretation of such data complex. As a result, many workers have either used artificial streams, or have manipulated the pH and other water quality variables of real streams, in order to i) quantify chemical changes more accurately, ii) reduce the number of uncontrolled variables, iii) produce greater or temporally different changes in pH or other variables than occur under field conditions and/or iv) manipulate the species of organisms being tested. It should be pointed out, however, that it is enormously difficult to control and adequately measure even the most important chemical variables in mesocosms such as artificial channels. Thus the results of such experiments are not always reliable or strictly comparable.

In a low-order stream in Michigan, USA, Maurice *et al.* (1987) studied the effects on the periphytic community (i.e the attached algae) of lowering the pH to 4 by addition of H_2SO_4. The authors postulated that the decreased rate of colonisation by periphyton was a consequence of reduced decomposition rates and nutrient availability at low pH. They also found that the species composition of the community altered, blue-greens (which are sensitive to acidity) largely being replaced by acid-tolerant genera. Maurice *et al.* concluded that alterations of the type they saw would have effects up the food chain. Similar experiments using enclosed mesocosms in a lake in West Virginia, USA (Havens & DeCosta 1987), examined the effects of lowering pH and adding Al. Again, the species composition of the algae (phytoplankton in this case) changed in the mesocosm in which pH had been lowered (to 4.7), while levels of chlorophyll *a* decreased on the addition of aluminium (50-180 $\mu g\ l^{-1}$). The species of zooplankton did not change at lowered pH values (they were already part of an acid-tolerant community) but zooplankton abundance decreased. The data generated by Havens & DeCosta indicated that the ability of some species to withstand decreased pH levels was increased with increased Al concentrations. Similar results were obtained and similar conclusions reached by Mulholland *et al.* (1986) for high-elevation streams in the Great Smokey Mountains, Tennessee, USA and by Genter & Amyot (1994) for artificial streams in Vermont, USA.

Results from several studies (see Herrmann *et al.* 1993) have measured slight increases in algal biomass as a result of acid precipitation. An suggested explanation (e.g. Sangfors 1998) is that the increased nitrate in the precipitation has a fertilizing effect, although Genter and Amyot had similar results for mesocosms acidified with sulphuric and not nitric acid.

Zische *et al.* (1983) examined the effects of lowered pH on benthic community structure using three artificial channels, each with pH controlled to a different level: pH 8 (ambient), pH 6 and pH 5. Both densities and diversity of invertebrates were lowest in the channel at pH 5. Damselflies, isopods and leeches were most tolerant of lowered pH, while snails and some amphipods were most sensitive. Spawning and development of embryos of the fathead minnow (*Pimephales promelas*) were inhibited only in the channel at pH 5.

Hall *et al.* (1987) simulated acidified snowmelt by adding HCl and $AlCl_3$ in separate experiments to the upper reaches of a small stream in New Hampshire, USA. They found that when the pH of the stream was lowered to 5.25-5.5 by HCl alone (ambient Al 15 µg l^{-1}), the drift of aquatic invertebrates was not altered; but at the same pH, when the concentration of Al was manipulated to 280 µg l^{-1}, invertebrate drift increased significantly. They concluded that not only increased but also fluctuating levels of Al may increase the biological stress of low-order streams.

A similar study (Bernard *et al.* 1990) on a British Columbian stream in Canada showed that invertebrate drift increased three-fold when pH was decreased from ambient (pH 7.0) to 5.9, while further manipulation by addition of Al (0.71-0.95 mg l^{-1}, pH 5.9), resulted in a six-fold increase in drift. The effects of lowered pH and increased levels of Al were also examined by experimental acidification and addition of Al in a Welsh stream by Ormerod *et al.* (1987). Once again, drift increased more significantly in the "Al zone" than in the "pH zone", while a species of mayfly, *Baetis rhodani*, declined in numbers, but only in the "Al zone". Brown trout (*Salmo trutta*) and salmon (*Salmo salar*) showed greater mortality in the "Al zone" than in the "pH zone".

Hall (1990) experimentally reduced the pH of a forest stream in Ontario, Canada, in order to examine the effects of short-term (4-8 days) fluctuations in pH. Such fluctuations are characteristic of streams receiving snow-melt: Gjessing *et al.* (1976) have shown decreases in pH from 6 to 4 during spring storms in Scandanavia (see also Kahl *et al.* 1992). Hall dropped the pH of the experimental stream from about 6 to 4.2 and found that the concentrations of both Al and Ca increased as a result. He found that the density (i.e. numbers of individuals per unit area) and the number of genera of benthic invertebrates decreased and, while the total number of invertebrates drifting did not increase, the number of mayfly nymphs did. He showed that discharge was the primary correlate of drift in spring, while Ca and dissolved organic carbon were the primary correlates in autumn. He concludes that most of the benthic invertebrates in his study area are adapted to sudden change during times of snowmelt and are thus not particularly sensitive to short-term decreases in water quality.

Kratz *et al.* (1994) looked particularly at the effects of pulses of acidified water on invertebrate drift in a stream in the Sierra Nevada mountains in California and found that invertebrate drift increased while the densities of benthic invertebrates decreased. Interestingly, effects were least pronounced for chironomid larvae, perhaps because they most of them could burrow into the substratum and thus avoid the pulses of low-pH water.

McCahon & Poulton (1991) dosed a soft-water stream in Wales with sulphuric acid, aluminium sulphate and limestone to produce a zone of Al at low pH, and one of Al, low pH and added lime. Each episode of dosing lasted about 24 hours. They found that the amphipod *Gammarus pulex* was killed more readily in the absence than the presence of lime although more died or exhibited behavioural alterations some time after dosing even when lime had been added.

While blackwater systems in the south-western Cape of South Africa are naturally acidic and commonly reach pH values as low as 4.5 (e.g. Day & King 1995), those in the continental USA are often circumneutral. Experimental addition to a blackwater Virginian stream of the strong mineral sulphuric acid reduced ambient pH values from 6.6 to about 4.5 (Smock & Gazzera 1996). The manipulation increased ambient Al concentrations, invertebrate drift and benthic microbial respiration but had little effect on the rate of decay of leaf litter or the densities or biomasses of benthic invertebrates; it would seem that the dissolved organic material in the stream was able to sequester some of the Al ions and make them unavailable to the biota (see section 6.5.1).

6.6.3 Field studies

Direct studies of polluted rivers provide more realistic information than mesocosm studies do on the effects of alterations in water quality on community and ecosystem structure but the interpretation of the results is often questionable, largely because antecedent conditions are seldom known precisely. Herrmann *et al.* (1993) provide an excellent review of what was known at the time of the effects in streams of what they call 'acid-stress' on primary production, invertebrates, fish, biotic communities and stream ecosystems. Little seems to have been published since that time.

6.6.3.1 Effects on stream communities

One of the few studies on the effects of alkalinization (rather than acidification) on stream communities was conducted by Morgan (1987) on streams in the Pine Barrens of New Jersey, USA. This area is underlain by sandy, nutrient-poor, acidic soils that have become less acidic as a result of agricultural and residential developments. Morgan showed that, as might be expected, species diversity of the periphytic communities was significantly higher in the more disturbed (i.e. more alkaline) streams, where nutrient levels (particularly nitrates) were also elevated.

Numerous studies have been performed on the effects of lowered pH on benthic invertebrate communities (see Hall 1990 for references). It has not always been possible to distinguish specifically the effects of lowered pH, although acid streams do tend to have reduced numbers and diversities of invertebrates. For instance, Guerold & Pihan (1989) and Guerold *et al.* (1991) examined fourteen streams in the Vosges Mountains in France. These streams drain forested catchments in which decline of coniferous forests has been attributed to acid precipitation. Acidification and increased concentrations of aluminium were found to occur at least during snowmelt and rainstorms. Diversity and abundance of benthic macroinvertebrates decreased with increasingly acid conditions: mayfly and plecopteran nymphs and caddisfly larvae dropped from 93 to 29 species, while their abundance was a third of that in the least affected stream. Ephemeropterans were the most sensitive, none occurring below a pH of 5.9. Some species of trichopterans were able to survive, and the benthic fauna of the most acidified stream was dominated by plecopterans. Similar results were obtained by by Simpson *et al.* (1985), working on headwater streams in the Adirondack Mountains in New York State; by Griffith *et al.* (1995) for headwater streams in the central Appalachian Mountains of West Virginia, both in the USA, and by Vuori (1995) for hydropsychid larvae in Finnish streams.

In contrast, the benthic invertebrates, including ephemeropterans and trichopterans, are speciose in, and often narrowly endemic to, south-western Cape rivers and wetlands (Wishart & Day 2002) where pH values naturally occur in the range 4.5-6.0 (e.g. Britton 1991, Dallas *et al.* 1998). Values may fluctuate over more than one pHunit seasonally and even diurnally (see Figure 6.2).

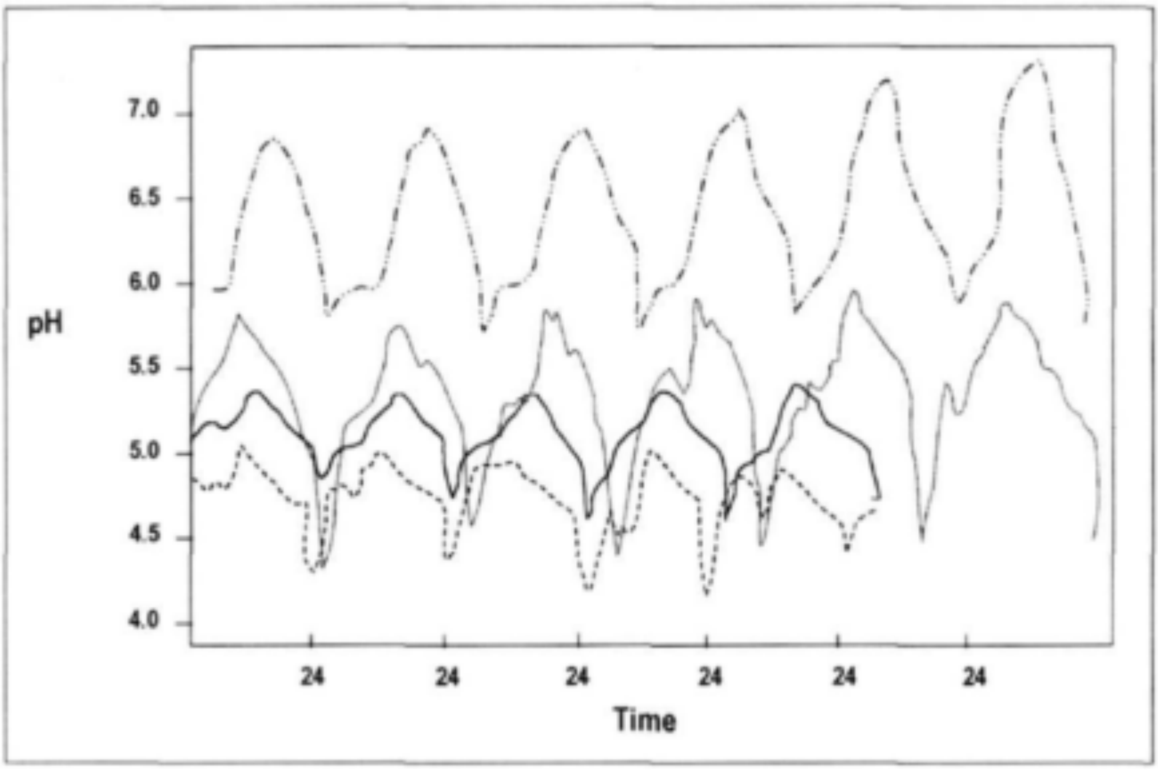


Figure 6.2 Diel variation in pH at two sites in mountains stream (—— ; - - - -) and two in foothills (········ ; — · · ·) of the south-western Cape rivers (modified from Dallas *et al.* 1998).

Carline *et al.* (1992) specifically examined the responses of fish, particularly brook trout (*Salvelinus fontinalis*), to episodic stream acidification in the Appalachian Mountains in Pennsylvania, USA. They found that when the pH dropped to about 4 during such episodes, concentrations of Al reached toxic levels, a peak value of 0.75 mg l^{-1} being recorded. During these periods, radio-tagged trout moved downstream.

Hall & Ide (1987) examined the change in community structure of benthic macroinvertebrates between surveys conducted in 1937 and 1985 in streams in Ontario, Canada. They found that the same taxa were present at those sites where pH depression in spring (as a result of snowmelt) are small (pH dropping from 6.4 to 6.1), while acid-sensitive species had disappeared from streams where pH depressions were large (pH dropping from 6.4 to 4.9). They concluded that the poorly buffered surface waters of these streams had become acidified some time within the previous four decades.

Collier & Winterbourn (1987) reported on studies on naturally acid and alkaline streams in New Zealand. They found that concentrations of Al were higher in the acid than the alkaline streams and that the density of invertebrates and the number of taxa was lower. They concluded that reduced diversity and density were probably due to reduced food supply and altered species composition of the periphyton (the major food) in the acid streams.

Szczesny (1990) described a number of acidified (pH 3.3-5) streams in a National Park in central Poland and noted that between pH values of 4 and 5, only stonefly nymphs and caddisfly and dipteran larvae were present; only larvae of the caddisfly *Limnephilus coenosus* occurred in streams with pH values <4.

6.6.3.2 The effects of lowered pH on food quality

It has long been known that severe acid pollution reduces, and eventually entirely prevents, aquatic decomposition of organic matter such as leaves, which form the base of the food chain in most mountain streams. Some direct experimental evidence comes from a study by Groom & Hildrew (1989), who examined the conditioning (i.e. the initial process of decomposition that makes leaves more suitable as food) of tree leaves in acid and circumneutral streams. They conclude that C:N ratios were higher (i.e. of poorer quality as food) in leaves from the acid stream, while the concentration of Al in the leaves was greater, so that in all, food quality was negatively affected by acidic conditions. Data generated by Haramis & Chu (1987) provided direct evidence that both survival and growth were impaired in of American Black Duck ducklings raised on acidified wetlands.

6.6.3.3 Recovery and rehabilitation of acidified streams

Attempts have been made over the years to restore acid-impacted streams, usually by the addition of lime. Results of various studies are too variable to allow generalisations. Fjellheim & Raddum (1992), for instance, report on recovery of the River Audna, in southern Norway, which has been limed at a rate of 8-11 t d^{-1} since 1985. (Fjellheim & Raddum give no discharge values for the river.) Acid-sensitive species of mayfly and stonefly nymphs, and caddisfly larvae, were first recorded two years after liming began. According to these authors, the reduction in physiological stress at higher pH was probably more important in encouraging the return of these species than changes in the food web that make organic detritus more accessible.

Lacroix (1992) also reports increases in numbers of both Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*) after dumping a single mass of 20 t of crushed limestone onto the bed of Fifteen Mile Brook in Nova Scotia. The effect persisted for at least three years even though the increase in pH was <1 unit.

In contrast, Diamond *et al.* (1992) examined the effects of liming agricultural land on the chemistry and aquatic fauna of the River Esk in north-western England. These authors found that the biota recovered to some extent despite the fact that liming seemed not to have affected water chemistry, in that rain water and stream water were chemically almost identical. On the other hand, Soulsby *et al.* (1997) were unable to detect increases in abundance or diversity of mayfly species in a number of streams in the Cairngorm region of Scotland, where atmospheric acidification had been decreasing measurably over the previous ten years.

6.6.3.4 South African studies

Cholnoky (e.g. 1958), working in South Africa, was one of the first botanists to draw attention to the fact that diatoms exhibit very clear pH preferences. It is said that he could estimate the pH of any stream from examining the diatom community found there! Somewhat more recently, Harrison (1965) examined the effects of acidic gold- and coal-mine effluents on aquatic biotas of streams. He showed that the rate of neutralization depended on geology, the pH of those polluted streams running over carbonate-rich formations rising much more rapidly than that of streams running over other formations. Both plant and animal species were reduced in the most acidic stream (at Witbank).

6.7 CURRENT pH STANDARDS IN SOUTH AFRICA

The data discussed above indicate that:

- A change in pH from that normally encountered in unpolluted streams may have severe effects upon the biota.
- The extent of acidification or alkalinization is important in determining the degree of severity of the effects.
- Some streams are naturally far more acidic than others and their biotas are adapted to these conditions.

This being the case, it is clear that there is no one pH or pH range that is suitable for all streams and water quality guidelines have to be site-specific with regard to pH and alkalinity. Since pH is a log scale, a change of one unit means a ten-fold change in $[H^+]$. Further, in poorly buffered waters pH can change rapidly (e.g. Figure 6.2). Thus no single guideline values can be set. In South Africa, water quality guidelines for pH require that the Target Water Quality Range be stated in terms of the background site-specific pH regime. In all cases, local background conditions (including diel and seasonal variability where appropriate) should be ascertained before a water quality objective is set for a particular aquatic ecosystem. Thus both spatial and temporal variability in pH need to be determined on a case- and site-specific basis. Spatial variability includes geographic and longitudinal differences (upper, middle and lower reaches of rivers and streams, for instance), while temporal variability includes diel and seasonal differences.

South African DWAF guidelines presently require for fresh surface waters that pH values should not be allowed to vary from the range of the background pH values for a specific site and time of day, by > 0.5 of a pH unit, or by >5%, and should be assessed by whichever estimated is the more conservative. It is clear, though, that even diurnal variations are often >1 pH unit (see Fig. 6.2). This being so, Dallas *et al. (1998)* provide a protocol for deriving guidelines based on natural fluctuations in pH from season to season and from year to year. It is important to remember, however, that where streams and wetlands are susceptible to the effects of acid rain, non-acidified reference conditions need to be used in deriving guideline values.

CHAPTER 7: CONDUCTIVITY, TOTAL DISSOLVED SOLIDS (TDS) AND MAJOR IONS

SUMMARY

Material dissolved in water is commonly measured as total dissolved solids (TDS), as conductivity, or as salinity. TDS represents the total quantity of dissolved material, organic and inorganic, ionized and un-ionized, in a water sample. Conductivity is a measure of the ability of a sample of water to conduct an electrical current. TDS and conductivity usually correlate closely for a particular type of water. Salinity refers to the saltiness of water. Natural TDS in rivers is determined by geological or atmospheric conditions. Anthropogenic activities such as industrial effluents, irrigation and water re-use lead to increases in TDS. Very little information is available of the tolerances of freshwater organisms to increased TDS. Generally it is the rate of change rather than the absolute change that is important. Juvenile stages are often more sensitive than adults and effects may be more pronounced in upper mountain streams, where organisms are generally not tolerant of stress. Ions most commonly found in natural waters are the cations calcium, magnesium, sodium and potassium, and the anions bicarbonate, carbonate, chloride and sulphate. Their characteristics and importance with respect to aquatic systems are discussed. The Target Water Quality Range for TDS is stated in terms of case- and site-specific TDS concentrations, taking into account background concentrations. TDS concentrations should not be changed by more than 15 % from the normal cycles of the water body under unimpacted conditions at any time of the year, and the amplitude and frequency of natural cycles in TDS concentrations should not be changed

7.1 INTRODUCTION

One of the major descriptors of the "quality" of a water sample is the total amount of material dissolved in it. This property of water is commonly measured in one of three ways: as total dissolved solids (TDS), as conductivity, or as salinity, all of which correlate closely in most waters. Whatever the measure used, the total amount of dissolved material determines both the human uses to which a particular type of water can be put and the biotic characteristics of aquatic ecosystems.

TDS, as its name suggests, is a measure of the total amount of soluble material in a sample of water. The greatest mass of this material in natural waters comprises inorganic ions. The commonest of these are usually the cations Na^+, K^+, Ca^{2+} and Mg^{2+} and the anions HCO_3^- (bicarbonate), CO_3^{2-} (carbonate), Cl^- and SO_4^{2-}. Together, these are often referred to as the "major ions" (see section 7.6.1.). Other, usually less common, inorganic ions include nutrients such as NO_3^- and PO_4^{3-} (see Chapter 10) and various species of trace metals such as iron, copper, aluminium and zinc (see Chapter 12). A small amount of inorganic material may also be present in un-ionized form (e.g. as hydroxides).

Many dissolved organic compounds are non-ionic. A major component of dissolved organic carbon (DOC) in many natural waters comprises a variety of chemically complex "humic" and fulvic acids. These substances, often together known as "polyphenolics", are the breakdown products of plant decay and, depending on the pH, are often present in ionic form. Thus the total number of ions in water may include organic as well as inorganic species.

TDS, then, represents the total quantity of dissolved material, both organic and inorganic, and both ionized and un-ionized, in a sample of water. It is usually measured by weighing the residue from a known volume of filtered water (mesh size 0.2-0.5 μm) evaporated to dryness at a temperature (<70°C) low enough to prevent volatilization of labile DOC. Units are usually quoted as mg l^{-1} or g m^{-3} (= parts per million). Measurement of very low quantities of TDS is often inaccurate because of the experimental errors involved in weighing small amounts of residue in large containers, while measurement of very high values is inaccurate because of the hygroscopic nature of some evaporites.

(Electrical) *conductivity* is another measure of dissolved material and is often used as a surrogate for TDS, for the following reason. Since the electrical conductivity of water is a function of the number of charged particles (ions) in solution, it is also a measure of the total quantity of salts, and therefore of total dissolved solids, in a sample of water. "Conductivity" in water quality terminology, is thus a measure of the ability of a sample of water to conduct an electrical current: the higher the conductivity, the greater the number of ions in solution. Conductivity is quoted as mS m^{-1} or, in non-SI terminology, as μS cm^{-1}, where S is a "Siemen", which is the reciprocal of an ohm (the unit of *electrical resistance). Since the majority of material dissolved in most water is ionic,* TDS and conductivity usually correlate closely for a particular type of water. For South Africa as a whole, DWAF personnel use the relationship

$$\text{TDS (mg l}^{-1}) = \text{conductivity (mS m}^{-1}) * 6.6$$

although a multiplicand of 5.5 is somewhat more accurate for naturally acidic south-western Cape waters.

A measure of conductivity does not include any un-ionized solutes. Thus in waters very rich in DOC, for instance, this relationship does not hold particularly well. Further, since the molar conductivity (the conductivity imparted to water by a mole of a particular solute) varies from ion to ion, slight variations will occur in the relationship depending on the proportions of the major ions in solution. Despite these disadvantages, conductivity is a more commonly quoted attribute of water than is TDS, both because conductivity is easy to measure accurately and because inexpensive, portable meters are available.

The term *salinity* is often used to refer to the saltiness of water. It was originally derived from the concentration of Cl^- in sea water (in which the proportions of the major ions are constant) but is now usually given as a dimensionless number (i.e. with no units) defined by reference to the electrical conductivity of sea water and measured as conductivity. Until recently, though, salinity was defined as the mass (in grams) of the dissolved inorganic solids in 1 kg of sea water and the unit was often given as ‰ (parts per thousand). Sea water has a salinity of 35, or 35‰, or 35 g l^{-1}. Since the quantity of dissolved organic matter in sea water is very small relative to the amount of inorganic matter, salinity and TDS are virtually identical in sea water. In fresh water, however, the proportion of dissolved organic matter may form a significant fraction of the whole. In this case, values for salinity and TDS may vary quite significantly for a single sample. For this reason, the term "salinity" is not commonly used for inland waters other than for very salty ("saline") lakes and pans. (For a more detailed discussion on TDS, conductivity and salinity see, for instance, Day 1990.)

In the following section, the terms "saline" and "salinity" are used only in the general sense of "salty" and "saltiness" and not with reference to the method of measuring that property.

7.2 TDS AND AQUATIC ENVIRONMENTS

7.2.1 Sources of salts

The majority of ions in most natural waters derive from weathering of the rocks over which they flow or from which they drain. In South Africa, for instance, many of the rocks of the Table Mountain Series are derived from ancient (*ca* 500-320 My) weathered sandstones that contain very little leachable material. In consequence, the waters draining these rocks are low in TDS. Rocks of the adjacent Malmesbury Shales were laid down as muds in ancient seas and still contain considerable quantities of leachable ions (Tankard *et al.* 1982). The waters draining these rocks usually have markedly higher TDS concentrations than those draining TMS rocks (Day & King 1995).

The other major source of ions is the atmosphere. The amount of water-soluble material in continental air is rather small but maritime air can carry significant quantities of sea salt (see, for instance, Gorham 1955, Hutchinson 1975), often known as "cyclical salt". TDS values of coastal rivers and wetlands may be considerably elevated by the precipitation of cyclic salt. In some rivers in south-western Australia (e.g. the Blackwood River: Morrissy 1974), rivers may have higher TDS values inland than at the coast as a result of this and other phenomena. The relative influences of atmospheric and geological sources of ions are discussed in, for instance, Gibbs (1970), Cornish (1987) and Day & King (1995).

7.2.1.1 Natural variations

TDS values for natural waters vary from about 1 mS m^{-1} (*ca* 7 mg l^{-1}) in some tributaries of the Amazon River (Day & Davies 1986) through 35 000 mg l^{-1} (ca 5000 mS m^{-1}) in sea water to about 330 000 mg l^{-1} (*ca* 50 000 mS m^{-1}) in brines in saline lakes reaching NaCl saturation point. The lowest values recorded for South Africa are about 0.9-3.6 mS m^{-1} (= 10-27 mg l^{-1}) (Waterkloof stream, Transvaal: DWAF data-base) and 1.8-3.1 mS m^{-1} (17-37 mg l^{-1}) (the Swartboskloof Stream near Stellenbosch: Britton 1991); the highest recorded value is that for Burgerspan (9790 mS m^{-1}, *ca* 65 000 mg l^{-1}) in the south-western Cape (Silberbauer & King 1991a; see also Day 1993). It is clear from these extreme values that there is no "ideal" TDS value for aquatic ecosystems generally. It is certainly true that most rivers and large lakes world-wide have TDS values <100 mg l^{-1} (mean volume-weighted average for rivers: Hutchinson 1975). In more arid areas, however, rivers may be naturally rather saline (e.g. Pillsbury 1981). A number of lower rivers in Israel, for instance, have mean TDS values around 3000-3600 mg l^{-1} (Herbst & Mienis 1985), while springs in the Mohave Desert in the USA maintain impoverished faunas at TDS values of 2400 mg l^{-1} (*ca* 388 mS m^{-1}) (Naiman 1976). Closer to home, a gypsous spring in the Namib Desert, with TDS values reaching 150 000 mg l^{-1} (conductivity 24 800 mS m^{-1}: Day & Seely 1988) maintains a limited but flourishing fauna and flora. One of the rivers in South Africa for which the highest salinities have regularly been recorded is the Sak River near Williston in the Karoo (maximal recorded values 7684 mS m^{-1}, 84 020 mg l^{-1}: DWAF data-base). Saline lakes and pans (TDS 3000 - 330 000 mg l^{-1}) are common even in presently mesic climates (e.g. parts of Canada) as well as in arid regions such as Australia, the Andes, China (e.g. Williams 1981) and parts of southern Africa (Seaman *et al.* 1991; Day 1993).

7.2.1.2 Anthropogenic influences

Human activities have severely increased the TDS concentrations of inland waters worldwide, particularly in arid regions (e.g. California: Caufield 1985; Australia: Schofield & Ruprecht 1989; Hart *et al* 1991; South Africa: du Plessis & van Veelen 1991; Flügel 1993). Apart from the obvious effects of discharging saline industrial effluents into rivers or lakes, increasing TDS levels (a process commonly known as *salinization* or mineralization) may be caused by irrigation, clear-felling and return of large quantities of sewage effluent to inland waters.

Irrigation causes salinization of rivers in two ways. On the one hand, although the irrigation water may be low in TDS, the water itself is taken up by crops or evaporates, leaving behind its solutes. Ion exchange processes in the soil, particularly in base-poor soils, result in accumulation of NaCl, which is washed out of the soil into rivers during rain. Secondly, irrigation may result in a rise in the water-table and subsequent evaporation from the surface of the now-wet soil (see e.g. Williams 1987; Hart *et al.* 1990, 1991). Clear-felling of large, deep-rooted trees in Australia (Hart *et al.* 1991) has also resulted in raised water-tables and subsequent salinization of the soils, and hence of rivers. Allison *et al.* (1990) have shown, for instance, that clearing of native vegetation in the Murray River Basin in south-eastern Australia has led to an increase in groundwater recharge by two orders of magnitude. Areas of south-western Australia are also becoming salinised (Day, pers. obs.) as a result of tree-clearing.

Re-use of water, whether it is recycled for immediate human consumption or returned to a stream and subsequently re-used further downstream, will result in partial evaporation during the cleansing process and will consequently lead to increased TDS levels (see, for instance, du Plessis & van Veelen 1991).

7.3 INFLUENCES OF INCREASED TDS LEVELS ON AQUATIC ECOSYSTEMS

Perhaps the two environmental variables that most decisively determine the communities of organisms living in a particular aquatic ecosystem are flow rate and salt concentration. Thus the faunas and floras of marine and inland waters are utterly distinctive, as, to a lesser extent, are those living in lakes as opposed to rivers. Saline or brackish inland rivers (as opposed to estuaries) are poorly known (e.g Williams *et al.* 1991) but numerous studies (e.g. Wood & Talling 1988; Hart *et al.* 1991) have shown that the individual species making up the faunas and floras of lakes and estuaries have very distinctive salinity tolerances.

Tolerance to TDS is species-specific. The physiological mechanisms involved are not addressed here but are discussed in reputable physiology textbooks (e.g. Schmidt-Nielsen 1990). An excellent review in relation specifically to water quality issues is to be found in Hart *et al.* (1991). Broadly speaking, most organisms have limited tolerances, being confined either to fresh waters (<2000 mg l^{-1}: Hart *et al.* 1991 or <3000 mg l^{-1}: Williams 1981) or to the sea (33 500 - 35 500 mg l^{-1}). Estuarine organisms, on the other hand, may have to survive changes in salinity from virtually zero to 35 000 mg l^{-1} over a single tidal cycle and therefore have very wide tolerance limits. The range of TDS values over which wetland organisms thrive may vary considerably. Some, such as the brine shrimp *Artemia salina*, are known from waters with TDS values between about a third of that of sea water (*ca* 12 000 mg l^{-1}) to crystallizing brines (*ca* 330 000 mg l^{-1}) (Schmidt-Nielsen 1990) but such very wide salinity tolerances are unusual.

7.4 TOLERANCES OF FRESHWATER ORGANISMS TO INCREASED SALINITIES

Very little information is available on the salinity tolerances of freshwater organisms. In general, it seems that many species are able to survive and even flourish at relatively high salinities. For instance, in their review, Hart *et al.* (1991) conclude that many freshwater blue-greens and bacteria can adapt readily to TDS values between 7000 and 14 000 mg l^{-1}.

Prinsloo & Pieterse (1994) have shown that increases in TDS concentration in the middle Vaal River have been accompanied by decreases in turbidity (probably as a result of flocculation). They have also shown that production in a green alga, *Monoraphidium circinale*, increased at TDS values between 500 and 2500 mg l^{-1}, while that of the diatom *Cyclatella meneghiniana* and the blue-green *Microcystis aeruginosa* were inhibited at these concentrations.

Studies have been performed on fishes of rivers threatened by salinization in order to determine the likely effects of increased ambient TDS levels on fish communities. Williams & Williams (1991) have examined the salinity tolerances of four species of native Australian fish fom the River Murray (present-day average TDS 500 mg l^{-1}) and found that the LD_{50} values varied from 20 800 to 43 700 mg l^{-1}, as long as the concentration was allowed to increase slowly over a number of days. Pimentel & Bulkley (1983) found that the upper preferred TDS ranges of three species of fish from the Colorado River (present mean annual TDS ca 1500 mg l^{-1}) varied between 4400 and 6600 mg l^{-1}. Thus in both cases it seems that the fish communities would be relatively unharmed by considerably increased salinities in both of these rivers. Occasionally papers have appeared on salt tolerances of freshwater invertebrate assemblages *in situ*. Magdych (1984), for instance, found slight changes in the mayfly assemblages of sites in a small stream in Oklahoma where salinity had increased.

A certain amount of salt in water can either protect aquatic organisms from, or sensitize them to, various pollutants such as heavy metals and biocides. Voyer & McGovern (1991), for instance, found that intermediate salinities (10 000 - 32 000 mg l^{-1}) protect the estuarine mysid (opossum shrimp) *Mysidopsis bahia* against cadmium exposure at cadmium levels between 1 and 9 μg l^{-1}, whereas at lower salinities cadmium becomes toxic at about 5 μg l^{-1}. In a similar study, Monserrat *et al.* (1991) showed that the estuarine Argentinian crab *Chasmagnathus granulata* survives better at low salinities (7500 mg l^{-1}) than at high salinities (30 000 mg l^{-1}) when exposed to the insecticide Parathion.

A number of generalizations can be made from these and other studies.

- It is often the *rate of change* rather than the final salinity that is most critical. Many organisms are able to adjust to slow change by a process of physiological acclimation that cannot be accomplished if an environmental change is rapid. (This is why toxicity testing usually includes both short-term acute and long-term chronic tests.)

- Juvenile stages are often more (or very occasionally less) sensitive to increased TDS levels than are the adults. For this and other reasons, it has been found that tolerances to increased salinities are often higher in the laboratory than in the field.

- In general, there seems to be a "critical level" of salinity at about 5000-8000 mg l^{-1} which marks the upper limit of survival of most salinity-tolerant freshwater animals. It is in this range that physiological processes are no longer able to acclimate any further and that chemical processes such as the loss of calcium by precipitation also become marked (Khlebovich 1990).

- Salinity may act as an antagonist or a synergist in relation to a variety of toxic pollutants. In general, salinity provides some protection against heavy metals, probably by modifying their chemical speciation.

- The responses of freshwater organisms to alterations in salinity are likely to be related to the evolutionary origins of the taxon of which they are part. Amphipods, for instance, are of marine origin and seem to be relatively insensitive to increased salinities compared to some insect taxa that may well have originated in fresh water.

- It cannot be emphasized enough that the above conclusions refer largely to species of middle and lower rivers, which are likely to be far less sensitive to increases in salinity than those living in the extremely pure waters of mountain streams and torrents. Unfortunately, virtually no information is available on salinity tolerances of these mountain stream organisms.

Even less information is available on the effects of increased salinities on whole communities or ecosystems. Brock (1985) has shown distinctive distribution patterns and degrees of species richness of wetland plants in Australian wetlands of differing salinity. On the other hand, Williams *et al.* (1991) were unable to show any relationships between macroinvertebrate community composition and salinity in two Western Australian rivers subject to salinization from agricultural practices. Nor were these authors able to find any similar studies reported in the literature. They did suspect, however, that *Hydridella glenelgensis*, a species of bivalve, may have become extinct in the Glenelg River as a result of increased salinities.

A good deal of circumstantial evidence suggests that alterations in distribution patterns of individual species or communities are a result of changes in salinity. For instance Dallas (1992) has recently compared the invertebrate fauna of the Berg River in the south-western Cape with that found in the early 1950s. *Afrochiltonia capensis*, a salt-tolerant estuarine species of amphipod (beach hopper), has increased in numbers in the salinizing lower reaches of the river, while the mayfly *Baetis bellus* has been replaced by another species, *Baetis latus*, probably also as a result of increased salinities in the reaches where it is now found. Further studies and analyses of existing South African data may be able to show both the effects of increasing TDS levels on rivers and the natural salinity tolerance of some riverine species.

7.5 CURRENT STANDARDS IN SOUTH AFRICA

Given that remarkably little information is available on salinity tolerances of aquatic organisms, and that conductivities in natural aquatic ecosystems vary so widely, no absolute values can be recommended and no national standards for preservation of aquatic life appear in the literature. The South African Water Act of 1956 required that TDS values for effluents should not exceed those of the intake by more than 75 mS m^{-1} for General Standard rivers and by not more than 15% for Special Standard rivers. This is still the legal standard but interim Water Quality Guidelines for aquatic ecosstems have been published. Because of potential differences in tolerance between species, and because of spatial differences in natural TDS, theTarget Water Quality Range is stated in terms of case- and site-specific TDS concentrations. In all cases local conditions should be determined, and variability due to seasonal changes need to be considered (see Dallas *et al.* 1998 for details).

For fresh surface waters:

- TDS concentrations should not be changed by more than 15 % from the normal cycles of the water body under unimpacted conditions at any time of the year, and
- the amplitude and frequency of natural cycles in TDS concentrations should not be changed.

7.6 INDIVIDUAL IONS

As well as having an effect on water quality by their contribution to TDS, the concentrations and proportions of the individual major ions also affect water quality. This section discusses the effects of each of the major ions on aquatic biotas and section 7.7. discusses the effects of the relative proportions of the major ions. See Gibbs (1970) and Day & King (1995) for more detailed discussions of the determinants of ionic proportions in inland waters.

7.6.1 The major ions

The ions most commonly found in natural waters are the cations calcium, magnesium, sodium and potassium, and the anions bicarbonate, carbonate, chloride and sulphate.

7.6.1.1 Calcium

Calcium is one of the major elements essential for living organisms. As well as being found as a structural material in, for example, bones, teeth, mollusc shells and crustacean (e.g. crab) exoskeletons, it is vital for muscle contraction, nervous activity, energy metabolism and a great variety of other biochemical interactions. Calcium ions are often the major cations in inland waters.

Although we know that calcium is a vital element, very little is known about the actual effects of changes in its concentration on aquatic biotas. It is clear, mainly from empirical evidence, that waters low in calcium may be unable to support molluscs and crustaceans, both of which require calcium for the construction of shells and exoskeletons (e.g. Beadle 1981), but even this is an overgeneralization. For instance in Lake Lungwe (Zaïre Basin), where conductivities vary between about 1.5 and 1.7 mS m^{-1}, and $[Ca^{2+}]$ is about 0.07 mmol l^{-1}, fish are entirely absent and snails with glassy, Ca-free shells are common (Marlier *et al.* 1955); in some extremely pure, virtually Ca-free waters of the Amazon, snail shells are composed almost entirely of the protein conchiolin (Sioli 1955).

7.6.1.2 Magnesium

Magnesium is an essential element, being found in chlorophyll and in a variety of enzymes and being involved in the processes of muscle contraction and the transmission of nervous impulses. Since it is usually found in relatively high concentrations, it is unlikely to act as a limiting nutrient or a toxin. Very little is known about its effects on aquatic organisms.

7.6.1.3 Sodium

Sodium is ubiquitous in natural waters and is the major cation in sea water and in many South African inland waters. It is the major cation involved in ionic, osmotic and water balance in all organisms and is also involved in the transmission of nervous impulses and in muscle contraction. Sodium is probably the least toxic metal cation (Hellawell 1986) and its effects on aquatic systems are almost entirely as a major contributor to TDS.

7.6.1.4 Potassium

Potassium, like sodium, is involved in ionic balance in all organisms, and in the transmission of nervous impulses and in muscle contraction in animals. Since it occurs in much lower concentrations than sodium does, potassium can sometimes act as a nutrient, the lack of which limits plant growth.

It has been shown (e.g. King *et al.* 1988) that potassium concentration tends to correlate most closely with invertebrate community structure in multivariate analyses of a number of streams. Because of the mathematical techniques employed, and the type of chemical data included, the significance of such results is questionable. The results do suggest, however, that potassium may act as a limiting nutrient for animal communities as well as for plants.

7.6.1.5 Bicarbonate and carbonate ions (see also section 6.2.2)

Bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}) ions, together with free CO_2, are the equilibrium products of CO_2 dissolved in water. They are usually expressed as alkalinity. The proportions of each depend on pH, so that at a pH below 5.4, all the CO_2 is in the free form; between pHs of 5.4 and 8.3, all the CO_2 is in the form of HCO_3^-; as the pH increases above 8.3, HCO_3^- becomes replaced by CO_3^{2-}; above a pH of about 8.8, only CO_3^{2-} is present. Bicarbonate ions are the dominant anions in many fresh waters, while carbonate ions may dominate in very alkaline waters. (See chapter 6 for further details of the bicarbonate-carbonate buffering system.)

Neither ion is toxic in itself. Indeed, since plants require carbon dioxide for photosynthesis, CO_2 and its equilibrium products can sometimes act as limiting nutrients. Conversely, under eutrophic conditions the photosynthetic activity of aquatic plants may shift the $CO_2/HCO_3^-/CO_3^{2-}$ equilibrium so far towards CO_3^{2-} that the pH may rise to 10 or more, which has other effects on the biota. Carbon dioxide, carbonate and bicarbonate ions form the major buffering system (i.e. resistance to change in pH) in most natural waters. Some waters (in the south-western Cape, for instance), virtually lack calcium ions and therefore have extremely low alkalinities because in the absence of calcium ions the carbonate/bicarbonate equilibrium tends towards free CO_2. They are thus both poorly buffered and low in pH. There is no doubt that some elements of the biotas of these systems seem to be confined to poorly buffered, acid waters but the extent to which water chemistry dictates their distribution is not yet known.

The carbonate/bicarbonate system assists in reducing the effects of the addition of acids and alkalis to fresh waters by its buffering action. Thus waters with high alkalinities are least likely to be affected by acid precipitation. Of course, when base cations like calcium and magnesium are removed from the system, then buffering capacity is lost and pH can decline rapidly.

7.6.1.6 Chloride ions and chlorine

Chloride is the major anion in sea water and in many inland waters, particularly in South Africa. Chloride ions are essential components of living systems, being involved in the ionic, osmotic and water balance of body fluids. Except where they have an effect by increasing the total dissolved solids, they exhibit no toxic effects on living systems.

Chlorine itself, on the other hand, is a gaseous element that dissolves in water to form hydrochloric acid, which is a strong acid that dissociates to form Cl^- and H^+ ions. Free chlorine in water is toxic and is often used as a disinfectant in swimming pools and in water purification works. Total residual chlorine concentrations of <1mg l^{-1}can significantly affect aquatic ecosystems (see, for instance, Steinman *et al.* 1992 for the effects of free chlorine on periphyton species composition and biomass). Chlorine can have further detrimental effects on water quality in that, in the presence of organic compounds, including humic and fulvic substances, it may form a variety of trihalomethanes such as chloroform (which is known to be carcinogenic in mice and rats) and carbon tetrachloride.

7.6.1.7 Sulphate

Sulphur in water occurs largely as the sulphate (SO_4^{2-}) ion. In living systems, sulphur is an essential component of proteins and is thus an essential element. In most natural waters, sulphate ions tend to occur in lower concentrations than either bicarbonate or chloride ions. Sulphates themselves are not toxic. In excess, however, they form sulphuric acid, which is a strong acid that reduces pH and can have devastating effects on aquatic ecosystems (see Chapter 6). This is particularly problematic in water seeping from mines, where sulphate levels can be extremely high (see section 19.2.1 on acid mine drainage). Sulphur dioxide, the gaseous precursor of sulphate, is a major component of acid precipitation. In regions where the rivers are poorly buffered, sulphate levels can increase significantly, causing sharp drops in pH.

In anoxic (oxygen-free) conditions, sulphate ions are reduced to hydrogen sulphide, which has a strong tendency to become oxidized and is thus an oxygen "scavenger". Hydrogen sulphide, or "bad egg gas", is therefore an indicator of reducing conditions. It is also toxic, inhibiting a number of enzymes important in cellular metabolism. Although numerous studies have detailed the toxic effects of hydrogen sulphide in the laboratory (see, for instance, Hellawell 1986), its effects in the field have not been well quantified.

7.7 PROPORTIONAL CONCENTRATIONS OF THE MAJOR IONS

The major ions of natural inland waters are derived from the rocks with which they are in contact, and from the atmosphere. Depending on the relative influences of these sources, different ions predominate in different systems (see sections 2.1. and 2.2.). Briefly, in South Africa the waters of the highveld tend to be dominated by calcium, magnesium and bicarbonate ions, whereas those of the coastal regions and the arid west tend to be dominated by sodium and chloride ions (Day & King 1995). Because the different proportions of ions result in differences in, for instance, buffering capacity and the availability of essential elements, they are of significance for the biota.

The relative proportions of the different major anions in water seem not to be of great biological significance, as long as a lack of bicarbonate does not limit plant growth.

The effects of differing cationic ionic proportions on the aquatic biota have often been implied, but are in fact rather poorly known. Bayley and Williams (1973) and Beadle (1981) briefly review the literature. Beadle points out that one might expect the effects generally to be insignificant, since aquatic animals are likely to have evolved in situations requiring efficient ionic regulation. Harrison *et al.* (1966) have shown, however, that the rate of egg-laying in the African freshwater snail *Biomphalaria pfeifferi* is reduced or halted at the high ratios of Mg:Ca found in some Zimbabwean streams. Claims have often been made that the ratio of monovalent to divalent cations can at least partly explain the distributions of some algae, although Shoesmith and Brook (1983) give evidence to show that such apparent relationships may be spurious. The only discussion on similar topics in southern Africa is to be found in Day (1993).

CHAPTER 8: DISSOLVED OXYGEN

SUMMARY

Most aquatic organisms are dependent on water for their survival. The maintenance of adequate dissolved oxygen concentrations is critical for the survival and functioning of aquatic biotas. Dissolved oxygen, as mg l^{-1} or percentage saturation, fluctuates diurnally, depending on the relative rates of respiration and photosynthesis of aquatic animals and plants. Factors causing an increase in DO include atmospheric re-aeration, increasing atmospheric pressure, decreasing temperature and salinity, and photosynthesis by plants. Factors causing a decrease in DO include increasing temperature and salinity, respiration of aquatic organisms, decomposition of organic material by micro-organisms, chemical breakdown of pollutants, re-suspension of anoxic sediments and release of anoxic bottom water. Generally, it is a depletion of DO that is observed in aquatic systems although super-saturation, i.e. in excess of 100%, may occur in eutrophic waters. The significance to aquatic biota of dissolved oxygen depletion depends on the frequency, timing and duration of such depletion. Continuous exposure to concentrations of less than 80% of saturation is harmful, and is likely to have acute effects, whilst repeated exposure to reduced concentrations may lead to physiological and behavioural stress effects. Generally, if the rate of change is rapid, adverse effects on the biota will increase significantly. The extent to which any organism is affected by a decrease in dissolved oxygen is determined by its dependence on water as a medium. The oxygen requirements of fish and other aquatic organisms vary with type of species (particularly warm- or cold-water species), with life stages (eggs, larvae, nymphs, adults) and with different life processes (feeding, growth, reproduction) and size. If possible, many species will avoid anoxic or oxygen-depleted zones. Juvenile life stages of many aquatic organisms are more sensitive than adults to physiological stress arising from oxygen depletion, and in particular to secondary effects such as increased vulnerability to predation and disease. Prolonged exposure to sublethal, low oxygen concentrations may lead to changes in behaviour, blood chemistry, growth rate and food intake. Many toxic constituents such as ammonia, cadmium, cyanide, zinc, etc, become increasingly toxic as DO concentrations are reduced. Current standards in South Africa use chronic and acute physiological effects on aquatic biota for assessing the effect of dissolved oxygen depletion on aquatic ecosystems. Criteria based on a Target Water Quality Range and Minimal Allowable Concentration use percentage saturation levels for protection of aquatic biotas. Site-specific modifications are applied if local conditions require that control be more or less stringent.

8.1 INTRODUCTION

Any organism that spends part or all of its life in an aquatic environment is dependent on the characteristics of the surrounding medium for its survival. One of the most important abiotic factors relating to the survival of most aquatic organisms is the concentration of dissolved oxygen in the water. Unlike air-breathing organisms, which are almost always guaranteed a readily available supply of air, most aquatic organisms are dependent on oxygen dissolved in the water. The maintenance of adequate dissolved oxygen concentrations is thus critical for the survival and functioning of aquatic biota. A few invertebrate species can survive by anaerobic metabolism for short or long periods (they are facultative anaerobes), however, and some bacteria are obligate anaerobes.

Oxygen is moderately soluble in water. Gaseous oxygen (O_2) dissolves into water from the atmosphere, and is also generated during photosynthesis by aquatic plants and phytoplankton. Equilibrium solubility, termed the saturation solubility, varies non-linearly with temperature, salinity and atmospheric pressure, and with other site-specific chemical and physical factors. Under natural conditions the concentration of dissolved oxygen fluctuates diurnally, depending on relative rates of photosynthesis and respiration by aquatic biota. DO is usually lowest near dawn, increasing during the day, peaking in the afternoon, and decreasing during the night. Seasonal variations arise from changes in temperature and biological productivity. In unpolluted surface waters, dissolved oxygen concentrations are usually close to saturation. Typical saturation concentrations at sea level, and TDS values less than 3000 mg l^{-1}, are: 12.77 mg/l at 5°C; 10.08 mg/l at 15°C; 9.09 mg/l at 20°C.

8.2 MEASURING DISSOLVED OXYGEN

Dissolved oxygen may be measured as milligrammes per litre (mg l^{-1}) or as a percentage of the saturation concentration at the time of sampling. It is important to distinguish between the two measurements. Concentration, in mg l^{-1}, is an important measure because it is the absolute amount of oxygen dissolved in the water and is the concentration that relates to the amount of oxygen that an organism requires, rather than the percentage saturation. Percentage saturation is the proportion of oxygen actually dissolved in water relative to the theoretical maximum calculated from tables, taking into account temperature, salinity and air pressure. Percentage saturation gives a useful estimate of biological activity. Results of less than 100 % of saturation concentration indicate that dissolved oxygen has been depleted from the theoretical equilibrium concentration. Results in excess of saturation (super-saturation of oxygen) can indicate eutrophication in a water body.

Dissolved oxygen concentration must be measured as the lowest instantaneous concentration recorded in a 24-hour period, or as the instantaneous concentration at 06h00 (DWAF 1996). If DO is measured at any time it is necessary to interpret the value relative to the time within the 24-hour cycle.

8.3 FACTORS MODIFYING DISSOLVED OXYGEN CONCENTRATION

The concentration of dissolved oxygen in water is affected by several factors (Table 8.1). Atmospheric pressure, temperature and salinity modify the rate at which oxygen is dissolved in water. Respiration by animals and plants, photosynthesis by plants, and aerobic decomposition of organic matter by micro-organisms affect the concentration of dissolved oxygen in the water.

Table 8.1 Factors modifying the concentration of dissolved oxygen (DO) in water.

Factors causing an increase in DO concentration
Re-aeration from the atmosphere (dependent on turbulence and oxygen deficit): more oxygen dissolves in water as atmospheric pressure increases
Low temperatures and salinities increase the solubility of oxygen in water
Photosynthesis by aquatic plants
Factors causing an decrease in DO concentration
High temperatures and salinities reduce the solubility of oxygen in water
Respiration of aquatic organisms (plants and animals)
Organic waste and aerobic decomposition of organic material by micro-organisms
Chemical breakdown of pollutants
Re-suspension of anoxic sediments or release of anoxic bottom water

8.3.1 Atmospheric re-aeration

Dissolved oxygen concentrations can be increased by natural diffusion of gaseous oxygen from the atmosphere into water, i.e. re-aeration. Re-aeration is influenced by water turbulence and the oxygen deficit of the water. Turbulence refers to the physical "activity" or movement of the water and depends on water velocity, depth and degree of exposure of the substratum at the surface. The greater the surface area of water exposed to the atmosphere, the greater the rate of re-aeration. Thus turbulent, swiftly flowing mountain streams will achieve greater re-aeration per unit time than will deeper, more sluggish lowland rivers. The oxygen deficit is the difference between saturation concentration (the calculated maximum concentration of oxygen in water at a particular temperature, salinity and pressure) and the actual concentration (ReVelle & ReVelle 1974). The rate of re-aeration increases as the oxygen deficit increases.

Atmospheric pressure, which decreases with increasing altitude, affects the solubility of oxygen in water. Oxygen solubility decreases as atmospheric pressure decreases. Based on these relationships, the solubility of oxygen should decrease with increasing altitude. It has long been believed that cold, high altitude mountain streams are more oxygen-rich than warmer lowland streams. This belief is based on the fact that oxygen solubility decreases with increasing temperature and altitude. It seems, however, that the combined effect of decreasing temperature and air pressure at higher altitude results in an almost constant oxygen concentration down the length of most rivers (Jacobsen 2000).

8.3.2 Temperature and salinity

The solubility of oxygen in water is inversely related to both temperature and salinity. Thus increased temperatures and salinities reduce the solubility of oxygen in water, decreasing the amount that can physically dissolve and hence be available to aquatic organisms. If organic loading is high, oxygen depletion is further accelerated by microbial activity, which can also increase temperature.

8.3.3 Photosynthesis and respiration by aquatic plants

Dissolved oxygen concentrations in water fluctuate diurnally in response to photosynthesis (light hours) and respiratory activity (dark and light hours) of aquatic plants and phytoplankton (Lloyd & Swift 1976). This may have serious implications for aquatic organisms inhabiting streams that are subject to nutrient enrichment and where large amounts of aquatic vegetation and/or phytoplankton are present. Decay of plant biomass can have the same effect as large inputs of organic matter, further reduction oxygen. Thus, excessive plant growth can deplete oxygen concentrations below the 4 mg l^{-1} limit required by warm-water fishes and result in fish kills (Cooper 1993).

8.3.4 Respiration by aquatic animals

The rate of oxygen uptake (respiration) by aquatic animals is affected by species, size, age, activity, physiological condition, nutritional status and level of stress (e.g. presence of toxins, temperature) of the respective organism (Hellawell 1986). For example, cold-water fish species generally die from oxygen stress at considerably higher concentrations of dissolved oxygen than do warm-water species (Boyd 1982). In a low-oxygen environment, fish react by increasing the rate of respiration, i.e. pumping more water over their gills (Cooper 1993).

8.3.5 Organic waste

The presence of oxidizable organic matter, either of natural origin (detritus) or originating in waste discharges, can lead to reduction in the concentration of dissolved oxygen in surface waters. Oxygen depletion may be due the aerobic decomposition of organic waste by micro-organisms or the breakdown of chemicals. The potential for organic wastes to deplete oxygen is commonly measured as biological oxygen demand (BOD) and chemical oxygen demand (COD). BOD is a common measure of organic pollution (see Chapter 9). COD is a measure of the oxidation of reduced chemical species in water, i.e. the "reducing capacity" of an effluent.

8.3.6 Re-suspension of anoxic sediments or release of anoxic bottom water

Under conditions of low flow, bottom water and sediments may become anoxic (i.e. an absence of free and bound oxygen). During periods of high flow such as floods or when a water body is dredged, re-suspension of these anoxic sediments may occur. Turnover or release of anoxic bottom water from a deep reservoir may also occur.

8.4 THE EFFECTS OF DISSOLVED OXYGEN CONCENTRATION ON AQUATIC BIOTAS

The concentration of dissolved oxygen in water may decrease or increase. The latter is termed super-saturation and generally indicates eutrophication in a water body. Generally, it is the decrease in dissolved oxygen in aquatic ecosystems that is observed and that may have adverse effects on many aquatic organisms (e.g. micro-organisms, invertebrates and fish), which depend upon oxygen for their efficient functioning. Of interest is the high variability of DO at small spatial scales (Kemp & Dodds 2001), indicating that sub-habitats for organisms vary spatially across distances of decimetres or less. DO concentrations are highly dependent upon local rates of photosynthesis, respiration and advective transport (Kemp & Dodds 2001).

8.4.1 Oxygen depletion

The significance to aquatic organisms of depletion in dissolved oxygen depends on the frequency, timing and duration of such depletion (DWAF 1996). Continuous exposure to concentrations of less than 80 % of saturation is most harmful, and is likely to have acute effects, whilst repeated exposure to reduced concentrations may lead to physiological and behavioural stress effects. Occasional short-lived depletion of oxygen is less important. In all cases, if the rate of change in dissolved oxygen concentration is rapid, adverse effects on biota will be increased significantly.

The extent to which any organism is affected by a decrease in dissolved oxygen is determined by its dependence on water as a medium. Most fish are 100% aquatic and are thus totally dependent on dissolved oxygen for respiration and are very sensitive to low concentrations (Alabaster & Lloyd 1980 cited by Hellawell 1986). The oxygen requirements of fish and other aquatic organisms vary with type of species (particularly warm- or cold-water species), with life stages (eggs, larvae, nymphs, adults) and with different life processes (feeding, growth, reproduction) (Alabaster & Lloyd 1982 cited by NWQMS 2000) and size (Olson & Rueger 1968). Cold water-adapted species such as Salmonidae (trout) are especially sensitive to depletion of dissolved oxygen. Reproduction and growth in these species is reduced under continuous exposure to oxygen concentrations below 100 % saturation. Juvenile life stages of many aquatic organisms are more sensitive to physiological stress arising from oxygen depletion than adults, and in particular to secondary effects such as increased vulnerability to predation and disease. If possible, many species will avoid anoxic or oxygen-depleted zones. Prolonged exposure of fish to sublethal, low oxygen concentrations may lead to changes in behaviour (Davis 1975), blood chemistry (Davis 1975), growth rate (Brungs 1971b), and food intake (Stewart, Shumway & Doudoroff 1967). Hypoxia (insufficient oxygen) may result in tissue damage, bleeding, increase in size of, and extreme loss of blood from, the gills, liver, kidneys and spleen of exposed fish (Drewett & Abel 1983). A few species, such as lungfish and some catfish, can, however, survive out of water for some time.

Certain insects, for example ephemeropterans (mayflies), plecopterans (stoneflies) and trichopterans (caddisflies), respire with gills or by direct cuticular exchange, and are thus subject to the same stresses as fish (Nebeker 1972). It has been proposed that invertebrates such as trichopteran larvae, ephemeropteran nymphs and plecopteran nymphs from highland streams are adapted to well-oxygenated conditions by having developed smaller gills that species from lowland streams (Illies 1964). A recent study has however, shown that, except for the trichopteran family Hydrophsychidae, taxa with small gills do not replace taxa with large gills as stream altitude increases and stream temperature decreases (Jacobsen 2000). Other insects are able to utilize atmospheric oxygen (e.g. many hemipterans and coleopterans carry air bubbles under their wings or under special mats of hairs: Hart & Fuller 1974) and are thus less affected by reduced dissolved oxygen concentrations. There is relatively little information on the effects of DO on aquatic insects. Nebeker *et al.* (1996) showed that the caddisfly, *Clistoronia magnifica*, could tolerate low DO concentrations for short periods, but that concentrations below 5 mg l^{-1} would cause reduced moulting success and delays in adult emergence.

8.4.2 Super-saturation

Super-saturation (i.e. > 100%) may occur in sluggish or stagnant waters, which discourage loss of oxygen to the atmosphere, as a result of oxygen production during photosynthesis by aquatic plants. Super-saturation may also be chemically induced. Super-saturation can cause gas bubble disease (oxygen bubbles surrounding the gills) and mortality in fish. Lethal effects mainly threaten non- or less-mobile life stages, such as eggs or fry.

8.4.3 Interactions with toxic constituents

Naturally occurring low concentrations of dissolved oxygen can be found in aquatic habitats such as wetlands with large accumulations of organic matter (e.g. Cooper 1993). In a low-oxygen environment, fish react by increasing the rate of respiration, i.e. pumping more water over their gills. When toxic pollutants such as ammonia, cadmium, cyanide, zinc, etc. are also present, an extra stress is created since the higher respiration rate means that more toxic substance is absorbed. In addition, many toxic compounds become increasingly toxic, as DO concentrations are reduced. For example, EIFAC (1973 cited by NWQMS 2000) reported that the acute toxicity of several common toxicants roughly doubled as the DO concentration was halved from 10 mg l^{-1} to 5 mg l^{-1}. Under anaerobic conditions in the water column or in sediments, heavy metals such as iron and manganese can appear in solution, as ferrous (Fe^{2+}) and manganous (Mn^{2+}) species, and toxic sulphides (S^-) may also be released.

8.5 CURRENT STANDARDS IN SOUTH AFRICA

Chronic and acute physiological effects on aquatic biota are the norms for assessing the effect of dissolved oxygen depletion on aquatic ecosystems (DWAF 1996). Useful measures of the impact of dissolved oxygen depletion are the 7-day mean minimum concentration and the 1-day minimum concentration:

- 7-day mean minimum concentration: the arithmetic mean of the daily (24-hour) minimum instantaneous concentrations measured over seven consecutive days, and
- 1-day minimum concentration: the lowest instantaneous concentration recorded in a 24-hour cycle, or the instantaneous concentration at sunrise

Criteria for dissolved oxygen concentrations (in terms of percentage saturation) are given in terms of the Target Water Quality Range (TWQR) and the Minimum Allowable Values (MAV) (DWAF 1996). These concentrations provide limits that will ensure protection of aquatic biota from the adverse effects of oxygen depletion. The MAV aims to protect sensitive life stages that may last for only a few days, and takes account the resilience of aquatic biota to short-duration oxygen depletion.

Table 8.2 Target Water Quality Range (TWQR) and Minimum Allowable Values (MAV) for dissolved oxygen concentrations in water bodies (DWAF 1996)

Criteria	Concentration	Condition	Application
TWQR	80% - 120% of saturation	6 a.m. sample or lowest instantaneous concentration recorded in a 24-hour period	Will protect all life stages of most southern African aquatic organisms endemic to, or adapted to, aerobic warm water habitats. Always applicable to aquatic ecosystems of high conservation value.
MAV	> 60% (sub-lethal)	7-day mean minimum	The 7-day mean minimum and the 1-day minimum should apply together. Violation of these minimum values is likely to cause acute toxic effects in aquatic organisms.
	> 40% (lethal)	1-day minimum	

Modifications can be considered on a site-specific basis where measurement of natural or un-impacted conditions indicates the TWQR to be too stringent or not appropriate. In particular, site-specific derivation of dissolved oxygen criteria that are different from the TWQR and the MAV should be considered in the cases of:

- naturally eutrophic systems;
- deep reservoirs where oxygen depletion or anoxia of the hypolimnion is a regular, usually seasonal, occurrence;
- where endemic or introduced organisms may have more stringent dissolved oxygen requirements, for example in cold-water habitats

Strict control should be applied to the frequency, duration and extent of oxygen depletion below 80 % of saturation, because of its rapid effect on aquatic organisms.

The following specific conditions should be applied (DWAF 1996):

- The expected range in dissolved oxygen must be determined by measurement at a site over several consecutive 24-hour cycles, repeated in each season to take account of climatic influences.
- Where natural physical, chemical or ecological processes lead to dissolved oxygen concentrations which fall below the MAV, then the minimum acceptable site-specific concentration should be set at a level which is related to the measured natural minimum concentration.
- Some upland reaches of southern African streams, in mountainous areas such as Lesotho and the Drakensberg range, are considered cold-water habitats, usually with high natural

dissolved oxygen levels. The dissolved oxygen requirements of organisms adapted or introduced into South Africa (such as trout) may be more stringent than the TWQRs.

- Key aquatic species of recreational, commercial or conservation importance may be identified, and site-specific guidelines should be derived according to the needs of these species.

CHAPTER 9: ORGANIC ENRICHMENT

SUMMARY

Dissolved and particulate organic matter is naturally present in aquatic ecosystems. Anthropogenically-derived organic discharges, which originate from or are produced by living organisms, may result in organic enrichment of the receiving water body. Major sources of organic enrichment include domestic sewage, food processing plants, breweries and vegetable canning, animal feedlots, abattoirs and cattle grazing. Of these, enrichment by organic matter from sewage and sewage effluents is probably the most common and extensively documented type of pollution in rivers. Most organic material in sewage is not directly toxic to aquatic organisms. The major effects of organic enrichment are a decrease in dissolved oxygen concentrations, an increase in turbidity and the concentration of suspended solids, an increase in nutrient concentrations and possible bacterial contamination of the receiving water body. Of these, reduced oxygen concentration, measured as Biological Oxygen Demand (BOD), is considered to have the most severe impact on aquatic biotas. In fact, organic waste is commonly referred to as "oxygen-demanding waste". Aquatic assemblages typically respond to organic enrichment through changes in species composition, increased densities of taxa tolerant to enrichment, and decreased densities or elimination of taxa sensitive to enrichment. Characteristic chemical and physical changes occur below the point of organic effluent input, together with changes in micro-organisms and macroinvertebrates. Both the duration and extent of the discharge (continuous versus episodic) and the river zone in which the enrichment occurs, influences the effect of the enrichment on the aquatic biota. Indicator species or taxa have been identified for most groups of organisms including bacteria, fungi, algae, protozoans and macroinvertebrates. Of these, macroinvertebrates are considered to be the best documented and understood indicators of organic enrichment.

9.1 INTRODUCTION

Dissolved and particulate organic matter (DOM and POM) are characteristically present in aquatic ecosystems, and detritivores in streams and in sediment communities of slow-flowing rivers depend upon POM for most of their energy (Moss 1980). The main difference between natural organic matter and organic enrichment relates to size, quantity and texture of the organic matter. Natural POM tends to occur in larger packets, like leaves, with a low surface-to-volume ratio, or is relatively refractory when finely divided, while anthropogenically-derived POM is usually soluble or finely divided and very labile (Moss 1980). Naturally occurring DOM is chemically diverse and may include a variety of high-molecular-weight refractory "humic" and "fulvic" compounds. DOM associated with pollution, on the other hand, tends to consist largely of smaller, biologically labile compounds. An organic discharge may be defined as one that is derived from or produced by living organisms. It is the most common type of enrichment occurring in aquatic ecosystems and is thus well documented (e.g. Hynes 1960, Hawkes 1979, Hellawell 1986).

9.2 SOURCES OF ORGANIC ENRICHMENT

Major sources of organic enrichment include domestic sewage; food processing plants, e.g. dairy and milk processing, breweries and vegetable canning; animal feedlots; abattoirs (Brungs 1971b, Hellawell 1986) and cattle grazing (del Rosario *et al.* 2002). Organic discharges have been referred to as "oxygen-demanding wastes" (Brungs 1971b) since depletion of dissolved oxygen is the primary effect of organic enrichment on receiving waters.

9.3 EFFECTS OF ORGANIC ENRICHMENT ON WATER QUALITY

Organic enrichment of aquatic ecosystems results in various chemical (dissolved oxygen, nutrient levels) and physical (turbidity and suspended solids) changes that in turn drive biological changes within the receiving water body. An organic discharge is not directly toxic to aquatic life but its effects may significantly change biotic community structure and biological processes. The main effects of organic enrichment are a decrease in dissolved oxygen concentrations, an increase in turbidity and the concentration of suspended solids, an increase in nutrient concentrations and possible bacterial contamination of the receiving water body.

9.3.1 Dissolved oxygen

The most noticeable consequence of organic enrichment is its effect on dissolved oxygen concentrations in water and sediments. Dissolved oxygen concentrations naturally fluctuate as a result of photosynthesis by plants (during the day) and respiration (day and night) of aquatic plants and animals (Lloyd & Swift 1976, and see chapter 8). On addition of oxygen-demanding wastes to a receiving water body, micro-organisms (bacteria, fungi, protozoans) use up dissolved oxygen while consuming or decomposing the waste. The rate at which this process occurs is measured as the biological oxygen demand (BOD). BOD summarizes the quality of all organics in a single value, thus making it an extremely useful measure of organic enrichment. BOD may be assessed by measuring the rate at which oxygen disappears from solution in a sealed bottle kept for 5 days in darkness at 20°C. Large amounts of organic matter, from animal wastes, for example, provide a greater BOD than an aquatic system can supply oxygen for, resulting in an "oxygen-sag". When organic matter exceeds the capacity of a system to assimilate it, a degradation cycle begins. The enhanced level of oxygen consumption by aerobic decomposers exceeds the rate of re-aeration and dissolved oxygen concentrations begin to fall. If DO decline continues, aerobic decomposers cease to function and anaerobic organisms populate the water and sediment.

BOD values of 2.3 mg l^{-1} and 2.6 mg l^{-1} were recorded in the unpolluted Praduik stream, Poland, (Dratnal & Kasprzak 1980) and an unpolluted stream adjoining the Opa River, Nigeria (Okoronkwo & Odeyemi 1985) respectively. Approximate biological oxygen demand values (5 day, 20°C) of typical effluents and fresh water are given in Table 9.1. Depletion of dissolved oxygen plays a major role in driving the biological changes (see Chapter 8).

Table 9.1 Biological oxygen demand values (5 days, 20°C) of typical effluents and fresh waters (From Hellawell 1986).

	Range of BOD (mg l^{-1})
Natural water - upland stream	0.5 - 2.0
- lowland stream	2.0 - 5.0
- large lowland river	3.0 - 7.0
Sewage effluent - crude sewage	200 - 800
- treated sewage	3 - 50
Livestock/agricultural waste - pig	27 000 - 33 000
- poultry	24 000 - 67 000
Abattoir	650 - 2 200
Meat packaging and processing	200 - 3 000
Fruit canning	635 - 2 100
Vegetable processing	480 - 4 400
Sugar beet	3 800 - 4 200
Sugar refining	210 - 1 700
Dairies - milk	300 - 2 000
- cheese	1 800 - 2 000
Breweries	500 - 1 300
Distilleries	> 5 000
Tannery	250 - 5 000
Textile waste	50 - 1 000
Pulp and paper manufacture	100 - 400
Petrochemicals	200 - 8 000

9.3.2 Turbidity and suspended solids

Turbidity and the concentration of suspended solids may increase when organic waste is discharged into a water body. The effects of these changes on aquatic ecosystems are covered separately (Chapter 5). In summary, an increase in turbidity and suspended solid concentration leads to a decrease in light penetration and primary production, which in turn reduces the availability of food to organisms higher up the food chain. Predator/prey interactions may be affected. Suspensoids that settle out may smother and abrade flora and fauna, alter the habitat, and lead to a change in benthic community structure. Deposition of organic sludge in slower-flowing water may lead to releases of methane and hydrogen (as hydrogen sulphide) if the organic matter decomposes anoxically. This extreme usually causes the elimination of normal benthic communities (Hellawell 1986).

9.3.3 Addition of nutrients

Nutrient loading is frequently associated with organic enrichment and compounds of nitrogen and phosphorus are normally present in large amounts in organic discharges. These nutrients are mineralised by decomposers, resulting in increased concentrations of dissolved and particulate nutrients in receiving water bodies. The partial biodegradation of proteins and other nitrogenous material can lead to elevated ammonia, nitrite (both toxic to many aquatic organisms) and nitrate concentrations (Hellawell 1986). More sophisticated wastewater treatment works are able to remove most of the nitrogen as N_2-gas and 70 to 95% of the phosphorus from domestic sewage in the form of bacterial sludges (Moss 1980). Details of the effects of nutrient enrichment are discussed in Chapter 10.

9.3.4 Bacterial contamination

Bacterial contamination of a water body may be caused by human waste discharged or entering a river as untreated sewage or from livestock waste. *Escherichia coli* is a non-pathogenic bacterium that occurs universally in the intestinal tracts of humans and many other mammals. Its presence in water bodies is used as an indicator of faecal pollution (Davies & Day 1998). It is likely that many South African rivers and wetlands, both urban and rural, are severely contaminated by faecal pathogens, particularly where informal settlements house poverty-stricken communities with no waterborne sanitation and meagre water supplies (Davies & Day 1998). Where water supplies are contaminated by *E. coli* and untreated, the bacterium may change its nature, causing health problems such as diarrhoea and gastroenteritis.

Faecal coliforms and faecal streptococci, whilst not pathogenic, are the primary indicators of the potential presence of pathogens (Larsen *et al.* 1994). Pathogenic organisms, when present in animal waste, can be transferred to humans via water (Diesch 1970 cited by Larsen *et al.* 1994). Some potential diseases that may be transferred to humans via cattle, for example, are salmonellosis, anthrax, turberculosis, tetanus, colibacilosus, etc. (Azevedo & Stout 1978 cited by Larsen *et al.* 1994).

9.4 EFFECTS OF ORGANIC ENRICHMENT ON AQUATIC BIOTA

Aquatic macroinvertebrate assemblages typically respond to organic enrichment through changes in species composition, increased densities of taxa tolerant to enrichment, and decreased densities or elimination of taxa sensitive to enrichment (Hynes 1960, e.g. Campbell 1978, Dratnal & Kasprzak 1980, Evans & Marcan 1976, Seager & Abrahams 1990, Szczesny 1974, Whitehurst & Lindsey 1990). For example, sewer overflows in Pendle Water, United Kingdom, lead to a reduction in macroinvertebrate densities, almost complete elimination of certain species (e.g. the isopod *Asellus aquaticus* and the mayfly *Baetis rhodani*) and an increase in oligochaetes, particularly tubificids (Seager & Abrahams 1990). Unpolluted water is characteristically represented by many species with few individuals within each species, whilst organically enriched water is represented by few species with many individuals in each. The polluted area may become dominated by taxa such as Chironomidae that are able to tolerate organically enriched water. For example, a multi-fold increase in the density of Chironomidae was observed after only 4 weeks of organic enrichment in an experiment simulating cattle grazing around headwater streams (del Rosario *et al.* 2002). Characteristic chemical and physical changes occur below the point of organic effluent input, together with changes in micro-organisms and macroinvertebrates.

Most biological changes in the receiving water body relate to the decrease in dissolved oxygen concentration resulting from the oxidative decomposition of organic waste. The duration of discharge, and the organic content of the effluent, will determine the severity of the effects on the aquatic ecosystem. For example, continuous low-level discharges lead to slow changes in oxygen levels, both spatially and temporally (Lloyd & Swift 1976), which may enable certain organisms to acclimate to the situation. High-intensity, episodic discharges of organic waste (e.g. overflow of sewage effluent during storm events) may have dramatic and instantaneous effects on biotic communities in the receiving waters (Seager & Abrahams 1990).

The effect of organic enrichment on aquatic ecosystems will also depend, to a certain extent, on the river zone in which it occurs. For instance, the biota of an erodible upper reach or mountain catchment zone is significantly more susceptible to organic enrichment than is a depositional zone (Hawkes 1979). The type and magnitude of flow, depth and substratum of the river will also affect the rate at which re-aeration takes place. Organisms associated with certain habitats, such as riffles, may be more susceptible to the effects of organic enrichment than sandy bottom dwellers (Hawkes 1979).

A classical study on organic enrichment, and one from which the saprobian system has been developed, is that of Kolkwitz & Marsson (1908, 1909 cited by Hellawell 1986). Kolkwitz & Marsson (1908 cited by Hellawell 1986) divided the river below the point of organic impact into four zones according to the processes taking place in each (Table 9.2).

Table 9.2 The saprobian system of zones for describing the effects of organic enrichment on aquatic ecosystems (after Kolkwitz & Marsson 1908, 1909 cited by Hellawell 1986).

ZONE	CHARACTERISTICS
Polysaprobic	Immediately below discharge: large quantities of rapidly decomposing matter (albumens, polypeptides and carbohydrates) reduce oxygen levels and may lead to anaerobic production of hydrogen sulphide, and hence unpleasant odour, with accumulation of black sludge deposits, bacteria and protozoans. Fauna severely restricted.
α-mesosaprobic	First stage of recovery: amino-acids are abundant, bottom muds no longer black, nuisance odour absent, bacterial counts still high. Fauna restricted.
β-mesosaprobic	Second stage of recovery: characterized by ammoniacal compounds of fatty acids; bacterial counts decline. Diversity of flora and fauna increases.
oligosaprobic	Full recovery: approaches upstream condition.

9.4.1 Indicator species or taxa

Different tolerance levels of species towards organic enrichment have led to the identification of indicator species or assemblages. Severe organic enrichment results in the development of profuse growth of "sewage fungus", a community of heterotrophic micro-organisms characteristic of organic contamination (Hellawell 1986). The community may include bacteria, fungi, algae and protozoans. Indicator species have been identified within each of these taxonomic groups, ranging from species intolerant of, tolerant of, and indifferent to, organic enrichment. The tolerant group may be further divided into those tolerant of mild, moderate and severe enrichment.

Macroinvertebrates, according to Hellawell (1986), include the species that are most documented and understood as indicators of organic enrichment. Certain macroinvertebrates, for instance larvae of the fly genera *Eristalis* (rat-tailed maggots) and *Psychoda* (sewage flies) are indicators of severe organic enrichment. As dissolved oxygen decreases and reaches zero the community becomes dominated by oligochaetes of genera such as *Tubifex* and *Limnodrilus*. As conditions improve midges such as *Chironomus* (bloodworms) occur. These are detritus feeders, consuming organic material in the waste, and burrowing into muddy and sandy bottoms and thus helping to facilitate re-oxygenation. Subsequent recovery of the biota depends to a certain extent on the faunal community characteristic of an undisturbed river within the same region and river zone. Generally ephemeropterans (mayflies), plecopterans (stoneflies), coleopteran larvae (beetles) and trichopterans (caddisflies) are indicative of relatively unpolluted water.

CHAPTER 10: NUTRIENT ENRICHMENT

SUMMARY

Various plant nutrients are required for normal plant growth and reproduction. It is nitrogen and phosphorus, however, that are most commonly implicated in excessive plant growth resulting from nutrient enrichment (eutrophication) of aquatic systems. Most nutrients are not toxic (exceptions include nitrite and ammonia), even in high concentrations, but when present in aquatic systems in these high concentrations, they may have a significant impact on the structure and functioning of biotic communities. Climatic and catchment characteristics influence initial nutrient concentrations in rivers. Anthropogenic sources of nutrients may be of the point-source type (e.g. sewage treatment works, industry, intensive animal enterprises) or nonpoint-source (e.g. agricultural runoff, urban runoff, atmospheric deposition) or urban runoff. Agricultural activities such as land clearing and fertilizer application are considered significant contributors to eutrophication of aquatic ecosystems. On entering an aquatic system, phosphorus is dissolved in the water column (as PO_4^{3-} ion) or adsorbed onto soil and other particles. High concentrations of phosphorus are likely to occur in waters that receive sewage and leaching or runoff from cultivated land. Nitrogen occurs abundantly in nature and is an essential constituent of many biochemical processes. Inorganic nitrogen may be present in many forms including ammonia (NH_3), ammonium (NH_4^+), nitrites (NO_2^-) and nitrates (NO_3^-). On entering aquatic systems, nitrates are rapidly converted to organic nitrogen in plant cells. Nitrite is an intermediate in the conversion of ammonia to nitrate, and is toxic to aquatic organisms. Un-ionized ammonia (NH_3) is also toxic to aquatic organisms and its toxicity increases as pH and temperature increase. Several management options are available for reducing the input of nutrients into aquatic ecosystems. Current water quality guidelines are designed to ensure that the trophic status of a water body does not change in a negative direction, e.g. from an oligotrophic to eutrophic state. The ratio of total inorganic phosphorus to total inorganic nitrogen is used to establish trophic status. Site-specific conditions need to be considered when calculating Target Water Quality Ranges. Guidelines for ammonia are given in the form of chronic and acute toxicity values.

10.1 INTRODUCTION

Plant nutrients are any elements required for normal plant growth and reproduction. In this sense, plant nutrients include carbon, nitrogen, phosphorus, potassium, calcium, magnesium, sulphate and silica, as well as other elements, termed "micro-nutrients", which are required in much smaller quantities (e.g. Addiscott *et al.* 1991). Of the major nutrients listed above, nitrogen (N) and phosphorus (P) are those most commonly implicated in excessive plant growth resulting from nutrient enrichment of aquatic ecosystems. P has been more widely implicated than N as the factor limiting primary production in freshwater systems (Hart *et al.* 1992, Correll 1998), although bioassay results suggest that both N and P can limit primary producers in streams (Dodds & Welch 2000). Nutrient enrichment is termed eutrophication and it can lead to an imbalance in biological communities, particularly to an increase in plant communities and associated water quality problems.

Traditionally, research on eutrophication has focused on standing water bodies as opposed to lotic systems. It is well known that as lakes age naturally, they become more enriched and progress from relatively unproductive (oligotrophic) to better-nourished, more highly productive (meso- or eutrophic) systems, which ultimately fill with organic material and become obliterated (Ruttner 1953 cited by Dye & Jones 1978). The system is normally in equilibrium over the short term. Accelerated primary production due to human-induced nutrient enrichment, however, leads to a change in this equilibrium. Eutrophication induced by man has been termed "cultural eutrophication" (Dye & Jones 1978). Most nutrients are not toxic (exceptions include nitrogen-derivatives ammonia and nitrite, and occasionally nitrate), even in high concentrations, but when present in aquatic systems in these high concentrations, they may have a significant impact on the structure and functioning of biotic communities.

This review focuses on nitrogen and phosphorus since concentrations of one or both of them lead to eutrophication of aquatic ecosystems. The other nutrients are dealt with in separate sections (See Chapter 7).

10.2 SOURCES OF NUTRIENTS

Certain factors naturally contributing to the provision of nutrients to any system are fixed and relatively constant with respect to a particular catchment or river reach. These include climatic factors (weathering, erosion, rainfall and variability of runoff) and catchment characteristics (surface geology and land form) (Table 8.1). Anthropogenic sources of nutrients may be of the point-source type (e.g. sewage treatment works, industry, intensive animal enterprises: Schofield *et al.* 1990) or non-point-source (e.g. agricultural runoff, urban runoff, atmospheric deposition: Carpenter *et al.* 1998). Point sources are relatively simple to measure and regulate, and can be controlled by treatment at the source (Carpenter *et al.* 1998), unlike non-point inputs which are diffuse and thus more difficult to measure and regulate. Of these, agricultural and urban activities are considered the major sources of phosphorus and nitrogen to aquatic ecosystems (Carpenter *et al.* 1998). Nonpoint-source inputs are often intermittent and linked to seasonal agricultural activities and irregular events such as heavy precipitation or major construction (Carpenter *et al.* 1998). Concentrations of total soluble nitrogen, inorganic nitrogen, total phosphorus and orthophosphate generally peak during the first two hours of runoff following a rainfall event (e.g. Simpson & Hemens 1978).

Associated with both agricultural, forestry and urban activities is catchment clearing, since clearing of indigenous vegetation from catchments is most often undertaken during preparation of land for agriculture or afforestation. Whatever the ultimate designated use of the land, the act of clearing, often impacts on the nutrient dynamics of the catchment and the associated water body. Pristine, forested catchments in Australia export little N and P and the predominant form of N is dissolved organic nitrogen (DON) (Harris 2001), which is presumed to be largely unavailable for algal growth (Smith and Hollibaugh 1997) and therefore contributes little to eutrophication. As catchments are cleared, exports increase and the predominant form of N changes from DON to dissolved inorganic N (DIN), mostly in the form of nitrate and ammonia, which is much more biologically available (Harris 2001). Exports of N from Australian catchments are similar to those from undisturbed temperate catchments but much less than those from undisturbed tropical catchments (Harris 2001).

Table 10.1 Major potential sources of nutrients (nitrogen and phosphorus)

Sources		
Climatic	Weathering of rocks and soil	
	Erosion	
	Rainfall	
	Variability of runoff	
Catchment characteristics	Surface geology	
	Land form	
Anthropogenic	Point sources	Sewage effluent
		Industrial discharge
		Intensive animal enterprises
		Detergents
	Diffuse sources	Agricultural surface runoff
		Disturbance of soil mantle
		Addition of fertilizers
		Manure
		Urban runoff
	Atmospheric deposition	Gases released from agriculture
		Burning of fossil fuels

10.3 MAJOR NUTRIENTS AND THEIR EFFECT ON AQUATIC ECOSYSTEMS

The major nutrients that contribute to eutrophication are phosphorus as phosphate ions (PO_4^{3-}) and nitrogen as nitrate (NO_3^-), nitrite (NO_2^-) and ammonium (NH_4^+) ions. Nitrogen and phosphorous may also occur in organic forms (especially nitrogen - in proteins) but are used by plants in the inorganic form. Microbes convert organic nitrogen and phosphorous to the inorganic form.

In standing waters (including natural lakes, impoundments, pool reaches of rivers with reduced flow) nutrient enrichment can lead to a rapid numerical increase in fast-growing plant and animal species which subsequently become pests and affect water quality for domestic, industrial, irrigation, recreational and other uses. Over the last three decades much research undertaken in lakes and reservoirs has allowed the prediction of algal biomass and other water quality parameters from nutrient loading and the concentrations of nutrients in the water column (e.g. Dodds *et al.* 1997). In contrast, general quantitative relationships between nutrient supplies and algal biomass in lotic systems are not well characterised (Dodds *et al.* 1997). Since this review is primarily aimed at assessing the effect of nutrient enrichment on riverine and wetland biotas, the wealth of literature pertaining to lakes is not referred to here.

Lotic systems are reported to be less susceptible to nutrient enrichment than are lentic systems (Porter 1975) because there is little retention in the moving water of rivers. The initial nutrient loading of a stream influences the effect that subsequent addition of nutrients will have on the system. This is clearly illustrated by the rate and extent of removal of orthophosphate (PO_4-P) and ammonium (NH_4-N) added to a nutrient-poor stream (McColl 1974). At some stage of nutrient enrichment a point will be reached where additional enrichment will be both visible and significant with respect to the aquatic biota. The longitudinal dynamic nature of a stream or river, unlike that of a lake, implies that for an effect to be apparent at any particular point, continual enrichment at that point needs to occur. It is therefore possible to reduce the effects of nutrient enrichment in rivers by removing or reducing the source of nutrient enrichment, given that the stream or river is flowing reasonably strongly.

Under conditions of low flow, however, a lotic system resembles a lentic system with respect to nutrient retention. Thus nutrient enrichment in slowly flowing riverine ecosystems may result in excessive plant (algal and macrophyte) growth (Cole 1973) and as the system becomes more productive, different species of algae may become more competitive and species composition may shift (Kelly 1998). This may lead to a change in macroinvertebrate biomass and altered communities (Bourassa & Cattaneo 1998). In extreme cases of primary production, organic carbon will build up in the system and cause a "low dissolved oxygen and a high pH" event (Dodds & Welch 2000). This may inhibit the growth of macroinvertebrates and fish and result in fatalities due to the reduced oxygen and elevated pH. Cooper (1993) considers oxygen depletion to be the greatest source of stress associated with eutrophication.

Blooms of cyanobacteria (blue-green algae) are also a prominent symptom of eutrophication and these blooms contribute to a wide range of water-related problems including summer fish kills, foul odours and unpalatability of drinking water (Carpenter *et al.* 1998). When the cyanobacterial blooms die or are ingested, water-soluble neuro- and hepatotoxins may be released, posing a serious and even life-threatening hazard to both livestock and humans.

10.3.1 Phosphorus

Phosphorus (P) has a major role in the structure of nucleic acids (eg DNA) and in molecules (eg ATP) involved in the storage and use of energy in cells (Addiscott *et al.* 1991). It occurs most commonly in dissolved form as the inorganic PO_4^{3-} ion. Soluble Reactive Phosphorus (SRP), i.e. immediately available phosphorus and phosphorus that can be transformed into an available form by naturally occurring processes, is seldom found in quantity (<0.01 mg l^{-1} P) in non-polluted water as it is utilized by plants and sequestered in cells. Knowledge of the role of processes and mechanisms that control the supply of bioavailable phosphate (i.e. orthophosphate and easily desorbed phosphate) is essential for the management of catchments, rivers and lakes to avoid eutrophication (Webster *et al.* 2001). Surface waters receive most of their P in surface flows rather than in groundwater since phosphates bind to most soils and sediment (Correll 1998). The phosphate ion is a highly surface-active species that is readily adsorbed to clay particles in the sediment and water column (Webster *et al.* 2001), or bonded to particles such as iron, aluminium, calcium or organic polyphenols (Addiscott *et al.* 1991). Thus during low-flow periods, sediments act as a sink for P entering the stream at high concentrations from point sources. Under high flow and/or anoxic conditions adsorbed P may be released from the sediments. Anoxia removes the adsorbtive layer of the clay particles and greatly increases pore-water concentrations (Webster *et al.* 2001). Processes in the phosphate cycle are summarized by the phosphate flow diagram (Figure 8.1) in which P-cycling is depicted together with the processes of inflow, outflow and sedimentation.

Higher concentrations of P are likely to occur in waters that receive sewage, and leaching or runoff from cultivated land. Import of P from agricultural land in New Zealand into aquatic systems is largely linked with soil runoff (McColl & Hughes 1981). Experimental studies (Meyer & Likens 1979; Mulholland *et al.* 1985 cited by Hill 1988) showed that P retention time in streams was affected by the flow regime and water residence time. Phosphorus (0.077 mg l^{-1}) added to experimental streams in Poland led to an increase in diatom biomass (Kownacki *et al.* 1985). Continuous enrichment of a pristine arctic river with 10 ppb of phosphate-phosphorus caused an immediate growth of attached algae for more than 10km downstream (Peterson *et al.* 1985). As a result of this increased photosynthesis there was an increase in bacterial activity in films on rocks and the system shifted from heterotrophy to autotrophy.

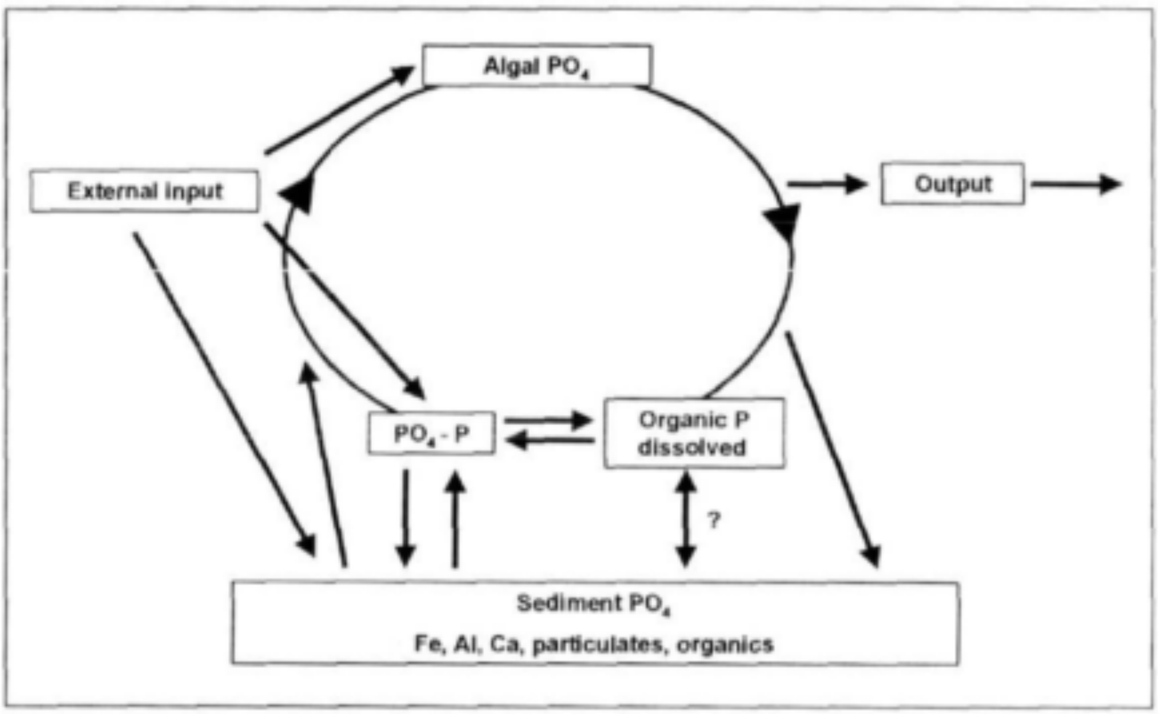


Figure 10.1 Phosphate flow diagram visualizing the recycling of phosphate through the biological compartments. (After Golterman 1975 cited by Golterman & Kouwe 1980).

10.3.2 Nitrogen

Nitrogen occurs abundantly in nature and is an essential constituent of proteins, which include the enzymes that catalyse all biochemical processes, and is therefore a major component of all living organisms. It also forms part of other essential constituents of cells: the chlorophyll that is essential for photosynthesis, the nucleic acids (DNA and RNA) in which the pattern for the organism's growth and development are encoded, the proteins, and the walls that hold cells together (Addiscott *et al.* 1991). Nitrogen-containing compounds are thus involved in practically all biological structures and biochemical processes.

In both natural and polluted waters, inorganic nitrogen may be present in many forms, but the ones that are measured by the common water quality tests include ammonia (NH_3), ammonium (NH_4^+), nitrites (NO_2^-) and nitrates (NO_3^-). A schematic representation of the nitrogen cycle is depicted in Figure 8.2. This cycle differs from the phosphate cycle in that nitrogen may enter and leave the cycle as gaseous N_2 by nitrogen fixation (e.g. blue-green algae such as *Anabaena* sp. and various species of Cyanobacteria have the ability to draw nitrogen directly from the air and incorporate it into the tissue) and by bacterial and chemical denitrification, which takes place in oxygen-poor conditions (e.g. sediments and heavily polluted rivers) (Golterman & Kouwe 1980).

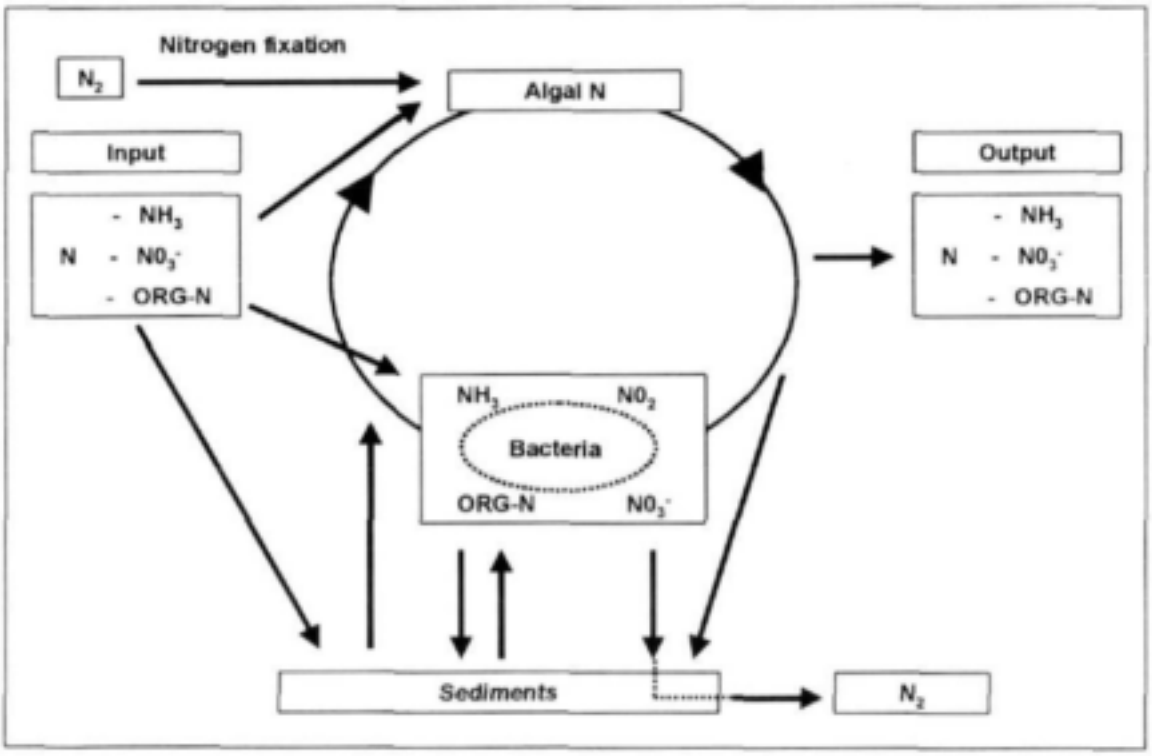


Figure 10.2. Schematic representation of the nitrogen cycle (From Golterman 1975 cited by Golterman & Kouwe 1980).

10.3.2.1 Nitrate

Nitrates are the end products of the aerobic stabilization of organic nitrogen (Figure10.2) and may enter water via fertilizers, agricultural runoff, etc. In spite of their many sources, nitrates are seldom abundant in natural surface waters (normally < 0.1 mg l^{-1} N), because photosynthetic action is constantly converting them to organic nitrogen in plant cells. They are, however, often found in high concentrations in ground water (Porter 1975). Nitrate is not normally toxic but high concentrations can be toxic to very young infants because NO_3^- binds with foetal haemoglobin to form a non-functional molecule, methaemoglobin.

10.3.2.2 Nitrite

Nitrite is a naturally occurring anion in fresh and saline waters. Human activities that increase nitrite concentrations in aquatic environments include industrial production of metals, dyes and celluloids, sewage effluents and certain types of aquaculture (Lewis & Morris 1986). It is the intermediate in the conversion of ammonia to nitrate (Figure 10.2) and is toxic at certain concentrations. Lewis & Morris (1986) reviewed the toxicity of nitrite to fish and described the mechanism of uptake and toxicity. Nitrite ions enter the fish through the chloride cells. From the blood plasma, nitrite diffuses into red blood cells, where it oxidises the iron in haemoglobin to the +3 oxidative state, resulting in the formation of methaemoglobin, which lacks the capacity to bind with oxygen. If the fish remains inactive (low oxygen demand) no immediate deleterious effects are experienced, but during activity (high oxygen demand) it may die of anoxia.

Toxic effects of nitrite are modified by water chemistry, particularly by chloride concentration (nitrite toxicity increases as Cl^- concentrations decrease); in fact there is an inverse linear relationship between nitrite toxicity and chloride concentration (Lewis & Morris 1986). Other anions, namely bicarbonate and nitrate, have also been shown to modify nitrite toxicity but not to the same extent as chloride. Some evidence suggests that calcium and pH modify nitrite toxicity, but it is inconclusive with respect to fish (Lewis & Morris 1986). A reduction in oxygen concentration will negatively affect fish exposed to nitrite since their blood carrying capacity is decreased.

10.3.2.3 Ammonia

Ammonia is a common pollutant generally associated with sewage and industrial effluents and occurs in either the free, un-ionized form (NH_3) or as ammonium ions (NH_4^+). It has been well established that the toxicity of ammonia is directly related to the concentration of the un-ionized form and that the ammonium ion has little or no toxicity (Williams *et al.* 1986), although it does contribute to eutrophication. In surface or ground water ammonium generally results from the decomposition of nitrogenous organic matter, and is one of the constituents of the nitrogen cycle (Figure 10.2) (McKee & Wolf 1963). Ammonia is present in small amounts in air, soil and water, and in large amounts in decomposing organic matter. Natural waters typically contain ammonia and ammonium compounds in concentrations below 0.1 mg l^{-1}.

Ammonia gas is readily soluble in water (solubility ≈ 100 000 mg l^{-1} at 20°C: Hart *et al.* 1992) and reacts with the water to form ammonium hydroxide. This then dissociates into ammonium and hydroxyl ions, which tends to raise the pH value of the water. The toxicity of ammonia and ammonium salts to aquatic organisms is directly related to the amount of undissociated ammonium hydroxide in the solution. At low to medium pH values, the ammonium ion dominates, but as pH increases ammonia is formed (Schubauer-Berigan *et al.* 1995) (Tables 10.2), the latter being toxic to aquatic organisms. Thus a high concentration of ammonium ions in water at low pH is not toxic, but if the pH is raised toxicity will develop. Ammonia toxicity is also affected by the concentrations of dissolved oxygen, carbon dioxide and total dissolved solids, and the presence of other toxicants, such as metal ions.

Un-ionized ammonia affects the respiratory systems of many animals, either by inhibiting cellular metabolism or by decreasing the oxygen permeability of the cell membrane (Gammeter & Frutiger 1990). Acute toxicity to fishes may cause loss of equilibrium, hyperexcitability, increased breathing rate, cardiac output and oxygen intake, and in extreme cases convulsions, coma and death (Hart *et al.* 1992). Chronic effects of ammonia include a reduction in hatching success, reduction in growth rate and morphological development, and pathological changes in tissues of the gills, liver and kidneys (USEPA 1986 cited by Hart *et al.* 1992). Behavioural observations on the mayfly *Ecdyonurus dispar* revealed that it increased ventilation of its gills following exposure to ammonia, indicating a respiratory effect (Gammeter & Frutiger 1990).

Table 10.2 Contribution of un-ionized NH_3 to total ammonia (expressed as a percentage), as a function of pH value and water temperature (After DWAF 1996).

pH	Water Temperature (°C)							
	0	**5**	**10**	**15**	**20**	**25**	**30**	**35**
6.0	0.0083	0.012	0.019	0.027	0.039	0.056	0.079	0.11
6.5	0.026	0.039	0.059	0.086	0.12	0.18	0.25	0.35
7.0	0.083	0.12	0.18	0.27	0.39	0.56	0.79	1.1
7.5	0.26	0.39	0.58	0.85	1.2	1.7	2.4	3.4
8.0	0.82	1.2	1.8	2.6	3.8	5.3	7.3	9.9
8.5	2.6	3.8	5.5	7.9	11	15	20	26
9.0	7.6	11	16	21	28	36	44	52
9.5	21	28	37	46	55	64	71	78

A study in Wales that assessed the impact of intensive dairy farming on river quality (Schofield *et al.* 1990) revealed significantly deleterious effects on macroinvertebrate communities in a stream subjected to 3-5 mg l^{-1} NH_3 (background level) and 20 mg l^{-1} (peak). The peaks were linked to washing of the farm area and rainfall events. Dense populations of a few pollution-tolerant taxa dominated receiving streams. Hermanutz *et al.* (1987) maintain that gradual increases in NH_3 concentration can lead to acclimation by fish populations, but that these populations were very susceptible to fluctuating NH_3 concentrations. NH_3 added to experimental streams (0.83 mg l^{-1}) resulted in a marked increase in ammonifying bacteria and microfauna, particularly ciliates (Kownacki *et al.* 1985). Under anoxic conditions ammonia is released into the water from the sediments (James & Evison 1979).

A recent study examined the chronic effects of ammonium and its products on three aquatic organisms, namely the amphipod, *Gammarus pulex*, the caddisfly, *Limnephilus lunatus* and the gastropod, *Radix ovata* (Berenzen *et al.* 2001). The study, performed in a microcosm at pH of 7 - 8, aimed to determine the no-observable effect concentration (NOEC) for the mixture, regardless of the breakdown of ammonium into ammonia, nitrate and nitrite. Of the three taxa, *G. Pulex* was the most sensitive. It exhibited a 98% decrease in abundance at an ammonium concentration of 3 mg l^{-1} (corresponding to 0.91 mg l^{-1} nitrite and 0.15 mg l^{-1} ammonia). This was 10-fold lower than studies in which the toxicity of substances formed by the breakdown of ammonium were investigated separately. *L. lunatus* and *R. ovata* were less sensitive and were affected at concentrations 10 times higher than *G. pulex*. This study highlights the synergistic effects that may occur when several toxic substances are present in a water body.

10.4 MANAGEMENT OPTIONS FOR REDUCING NUTRIENT ENRICHMENT

Reversal of eutrophication requires the reduction of P and N inputs (NRC 1992 cited by Carpenter *et al.* 1998). The following list provides information on management options for reducing nutrient enrichment in aquatic ecosystems.

1. Removal of nutrients from sewage. The treatment of domestic wastewater is fairly advanced and 70 - 95% of phosphorus, and most nitrogen, can be removed (Moss 1980).
2. Improvement of agricultural practices, including both land preparation and fertilizer application. Various studies have shown that a large percentage of nutrient enrichment directly or indirectly results from excessive fertilizer application (McColl & Hughes 1981, Carpenter *et al.* 1998). The following management options linked to agricultural activities have been synthesised from McColl & Hughes (1981) and Carpenter *et al.* (1998).
   - Selective and specific application of fertilizers to reduce direct entry into water bodies.
   - Timing of application to avoid heavy rainfall periods and to coincide with peak agricultural growth.
   - Avoidance of fertilizer application to areas susceptible to erosion e.g. overgrazed or trampled land.
   - The establishment of buffer strips in riparian zones to mitigate the movement of nutrients from adjacent steeper slopes and into water. Vegetated buffer strips reduced P transport to streams by 50-85% (Osborne & Kovacik 1993).
   - Management of wastes from high-density livestock operations as a point source of pollution.
3. Control of urban nonpoint-source pollution via the use of retention ponds, wetlands, greenways, litter control and street sweeping, reduction of impervious areas, and reduction of erosion, especially from construction sites.
4. Reduction of atmospheric deposition of N by more efficient use of fertilizers and improved handling of animal wastes.

10.5 CURRENT STANDARDS IN SOUTH AFRICA

10.5.1 Total inorganic phosphorus (TP) and total inorganic nitrogen (TN)

Water quality guidelines for the control of nutrient enrichment need to ensure that the trophic status of the water body does not move in a negative direction, i.e. to a more eutrophic state. It is therefore necessary to know both the natural trophic state of the system and the likely concentrations of P and N that may cause a shift in nutrient status. Management criteria based on TP and TN (i.e. total phosphorus and total nitrogen) are considered the best option for the control of nuisance algal growth in streams and river (Dodds *et al.* 1997) since instream TP and TN concentrations are more indicative of the nutrients that are ultimately biologically available for benthic algal growth than are DIN or SRP (Dodds *et al.* 1997). They considered target concentrations of TP and TN maintained below of 0.03 mg l^{-1} TP and 0.35 mg l^{-1} TN respectively should ensure chlorophyll a concentrations below 100 mg m^{-2}. Chlorophyll *a* is often used as a non-specific indicator of the trophic status of a water body (NWQMS 2000). These values may however vary from system to system, highlighting the importance of including some form of regional reference condition.

South African water quality guidelines (DWAF 1996) recommend the measurement of total inorganic phosphorus (TP) and total inorganic nitrogen (TN). The ratio of TN:TP may then be used to establish the trophic status of the water body. Unimpacted streams typically have an TN:TP ratio greater than 25-40:1; whilst most impacted (i.e. eutrophic or hypertrophic) systems have an TN:TP ratio less than 10:1. The TN:TP ratio is affected by turbidity since high turbidity can limit algal growth despite there being adequate nutrients available (NWQMS 2000).

Changes in the trophic status accompanied by the growth of algae and other aquatic plants in rivers, lakes and reservoirs is the norm used to assess the effects of phosphorus and nitrogen on aquatic ecosystems (DWAF 1996). Spatial and temporal variations in natural TP and TN concentrations need to be taken into account when assessing the potential effect of elevated concentrations on the aquatic environment, i.e. site-specific conditions need to be considered. Occasional peaks of inorganic phosphorus and inorganic nitrogen concentrations above the Target Water Quality Range (TWQR) are less important than continuously high concentrations. Single measurements of TP or TN are a poor basis for comparison. Average summer TP and TN concentrations provide the best basis from which to estimate the likely biological consequences of TP and TN. Weekly concentrations, averaged over a period of at least 4 weeks, should be compared with the TWQR.

The TWQR for all surface waters (DWAF 1996):

- TP or TN concentrations should not be changed by more than 15% from that of the water body under local unimpacted conditions at that time of year; and
- the trophic status of the water body should not increase above its present level, although a decrease in trophic status is permissible; and
- the amplitude and frequency of natural cycles in TP and TN concentrations should not be changed.

TP and TN concentrations below 0.005 mg P/l and 0.5 mg N/l respectively are considered to be sufficiently low that they can limit eutrophication and reduce the likelihood that nuisance growths of algae and other plants will develop. However, in the presence of sufficient available phosphorus, nitrogen-fixing organisms will be able to fix atmospheric nitrogen to make up for any deficit caused by low inorganic nitrogen concentrations. The information given in table 8.3 below illustrates typical symptoms associated with selected ranges of TP and TN concentrations.

Table 10.3 Symptoms or effects associated with selected ranges of inorganic phosphorus and nitrogen concentrations (After DWAF 1996).

Average Summer Concentration (mg/l)		Symptoms or Effects
Inorganic Phosphorus	Inorganic Nitrogen	
< 0.005	< 0.5	Oligotrophic conditions with no or very few water quality problems; usually low productivity systems with rapid nutrient cycling; no nuisance growths of aquatic plants or undesirable blue-green algal blooms.
0.005 – 0.025	0.5 - 2.5	Mesotrophic conditions with moderate or occasional water quality problems; usually productive systems; occasional nuisance growths of aquatic plants and blooms of blue-green algae; algal blooms seldom toxic.
0.025 – 0.25	2.5 - 10	Eutrophic conditions with frequent water quality problems; usually highly productive systems, with frequent nuisance growths of aquatic plants and blooms of undesirable blue-green algae; algal blooms may include species which are toxic to man, livestock and wildlife.
> 0.25	> 10	Hypertrophic conditions with almost continuous water quality problems; usually very highly productive systems; frequent nuisance growths of aquatic plants and blooms of undesirable blue-green algae, often including species which are toxic to man, livestock and wildlife.

10.5.2 Ammonia

Single measurements of ammonia are of limited use and it is recommended that weekly ammonia concentrations, averaged over a period of at least 4 weeks, with the minimum and maximum values reported, should rather be compared with the TWQR (DWAF 1996). Interpretation of the ammonia guidelines is based on the total ammonia concentration of a water sample, in addition to the proportion of the toxic un-ionized form. The potential effect of ammonia on the aquatic environment is modified by the chemical species present, the relative proportions of each and other factors such as pH, temperature and dissolved oxygen concentration.

The chronic and acute toxicity of ammonia are used as norms for assessing the effect of ammonia in the aquatic environment. Appropriate data are obtained from chronic and acute laboratory tests to set criteria for ammonia concentrations in the natural environment. The TWQR and criteria for un-ionized ammonia in aquatic ecosystems is given in table 8.4 (After DWAF 1996).

Table 10.4 Target Water Quality Range and criteria for un-ionized ammonia in aquatic ecosystems (After DWAF 1996).

TWQR and criteria	Un-ionized ammonia concentration (μg N/l)
Target water quality range (TWQR)	≤ 7
Chronic effect value (CEV)	15
Acute effect value (AEV)	100

CHAPTER 11: BIOCIDES

SUMMARY

Biocides are chemicals that kill living organisms and that are used in the control of pests, usually associated with agricultural crops and vector-borne diseases. The most commonly used biocides are herbicides, fungicides and insecticides. Potential sources of biocides in aquatic systems include direct application (for pest control), industrial effluents, sewage, leaching and runoff from soil, and deposition of aerosols and particulates. Studies have concentrated on biocide residues in the biotic and physical environment, bioaccumulation, determination of tolerance (acute and chronic) limits of aquatic organisms and the effects of biocides on whole communities. The nature, modes of action and toxicity of biocides vary considerably. Generally, organochlorine insecticides (e.g. DDT, dieldrin) are the most hazardous with respect to the natural environment and their use has thus been banned in many countries. These biocides are persistent in the environment, concentrating in organisms and thus through food chains. Methods for the detection and quantification of biocides are complex and expensive. Analyses are complicated by the small quantities of biocides found in water and the variety of breakdown products, with variable toxic properties, of most biocides.

11.1 INTRODUCTION

Biocides are chemicals that kill living organisms and are therefore used in the control of pests, often associated with agricultural crops and vector-borne diseases. Most biocides contaminating aquatic ecosystems are herbicides, insecticides and fungicides. Biocides enter aquatic environments by direct or indirect application (for pest control); in industrial effluents (including disposal of agricultural waste); in sewage; by leaching and runoff from soil; and by deposition of aerosols and particulates (Robinson 1973). Biocidal chemicals are usually categorized as inorganics (e.g. arsenicals, mercuricals, borates and fluorides), natural organics (e.g. rotenone, pyrethrum and nicotine) and synthetic organics (e.g. chlorinated hydrocarbons, organic phosphates and thiocarbamates) (McKee & Wolf 1963).

The era of synthesis and use of organic biocides began in 1940, and their production soon mushroomed: by 1973, Edwards reported approximately 1000 biocidal chemicals in common use around the world. Only in the late 1950s did the long-term ecological hazards of biocide use began to manifest themselves, however (Carson 1962). Since then much research has been directed towards determining the effects of biocides on both target- and non-target organisms. The effects of synthetic biocides on the biophysical environment are exacerbated by the fact that they may degrade to form breakdown products, some of which may be toxic, some persistent, and many difficult and expensive to measure. This results in problems in accurate measurement of quantities of biocides, and their effects, in natural systems (see 11.8.2).

Most ecological studies on biocides have estimated the concentrations of active ingredients and residues in the environment (e.g. McNaughton 1988 for ground water in Canada; Weaver 1993 for ground water in South Africa; Kimborough & Litke 1996, for Colorado streams; Liess *et al.* 1999 for a stream in Germany; Schulz 2001, Schulz *et al.*2001a, b for the Lourens River, Western Cape); bioaccumulation (e.g. Schmitt *et al.* 1983; Ohlendorf *et al.* 1988; Stendell *et al.* 1988; Barron 1990;

Grobler 1994; Roux *et al.* 1994); determination of tolerance (acute and chronic) limits of aquatic organisms (e.g. Phipps *et al.* 1995, and see Hellawell 1986 for a review); effects on fish behavious (Matthiessen & Logan 1984); the effects of biocides on biotic interactions or whole communities (e.g. Raven & George 1989; Davies & Cook 1993; Maloney 1995; Schulz 1998) and on ways of preventing or reducing contamination of aquatic ecosystems (e.g. Schulz & Peall 2001; Schulz *et al.* 2001c). Hellawell (1986) reviews the toxicity of various biocides to aquatic organisms.

11.2 ENVIRONMENTAL CONTAMINATION IN SOUTH AFRICA

The extent of environmental contamination by biocides has been intensively investigated in many parts of the world, although less is known about the situation in Africa, including South Africa, than in the developed world. While the extent of biocide contamination is virtually unknown for most of South Africa (for instance, practically no data exist for of large water storage reservoirs), a good deal of information is available for the south-western Cape. London *et al.* (2000) and Dalvie & London (2001), for instance, have investigated surface and ground water concentrations of chlorpyrifos and endosulfan in the Stellenbosch area, with regard to contamination of drinking water in rural areas. Concentrations of both insecticides were generally low, although most samples did contain measurable traces (or more) of both substances. Davies (1997) found significant traces of various insecticides in small farm dams in the south-western Cape, while Schulz and colleagues (e.g. Schulz 2001, Schulz & Peall 2001, Schulz *et al.* 2001a, b, c) have examined insecticide contamination of the Lourens River.

While DDT was banned for agricultural use in South Africa in 1976 (Curtis & Jenkins 2001) it is still used for mosquito control (du Toit 1995) because it is just about the only insecticide still effective in controlling certain species of malaria-carrying mosquitoes. (See section 11.5 for a brief discussion of resistance to pesticides.) Appleton (1985) provides a review of molluscicides used in South Africa for the control of the vectors of *Schistosoma*, the parasite that causes bilharzia; freshwater snails are the intermediate hosts of *Schistosoma* and attempts are commonly made to control them by chemical means.

11.3 TOXIC EFFECTS OF BIOCIDES IN AQUATIC ECOSYSTEMS

11.3.1 Laboratory and mesocosm experiments

The relative toxicities of different biocides to different organisms have been investigated both in the laboratory and (more recently) in the field but an understanding of these phenomena becomes complicated by the increasingly huge number of chemical compounds being developed and the variety of test organisms in use. Thus this review does not attempt to provide a complete account of different biocides or the extent of their toxicity. For detailed lists of toxicity values for numerous biocides the reader is referred instead to the web sites listed at the end of the report.

Extrapolation of toxicity test data to ecological systems is complicated by site-specific and species-specific variability, by the method of application, and by experimental conditions. Physical variables such as temperature, salinity, pH and dissolved oxygen affect bioassays and complicate comparisons. Because of these difficulties, Livingston (1977) proposed that acute toxicity testing be viewed merely as the preliminary step in determining the potential effect of a given biocide on aquatic organisms.

Further, Brown (1982 cited by Barlin-Brinck 1991) noted that sublethal effects of biocides, on behaviour, growth and reproduction, for instance, are not necessarily revealed by conventional LC_{50} tests. The situation is not significantly different today.

In an early attempt to relate laboratory data to the situation in the field, Hansen & Garton (1982) assessed the ability of a standard set of freshwater single-species toxicity tests (as per the USEPA guidelines) to predict the effects of the insecticide diflubenzuron (0.05 to 0.09 mg l^{-1}) on laboratory stream communities. The tests correctly predicted the concentrations of diflubenzuron that resulted in direct mortality of affected stream organisms but more subtle effects due to altered interspecific interactions were not predicted.

It is not normally possible to mimic field conditions adequately in a laboratory, while attempting to do any well controlled experiments in the field is seldom successful. A considerable amount of pesticide toxicity testing is therefore done in mesocosms, which are large artificial containers approximating in some degree to field conditions and often kept in the field. Schulz & Liess (1995) and Liess & Schulz (1996), for instance, examining chronic effects of short-term contamination with fenvalerate, lindane and parathion on caddisfly larvae, demonstrated slight decreases in emergence rates and increases in mortalities even at concentrations as little as 1-10 ng l^{-1} applied for one hour. These effects occurred at concentrations orders of magnitude lower than anything previously demonstrated. Examining sublethal effects of azinphos-methyl and fenvalerate in stream mesocosms, Schulz & Dabrowski (2001) have shown increased predation by fish on mayfly nymphs as a result of the pesticides causing the mayflies to drift. It is clear from these and other mesocosm studies that we still have a great deal to learn about sublethal effects in field conditions.

Some mesocsom experiments have shown that short-term exposure to herbicides does not necessarily have any measurable effects on stream biotas. Schneider *et al.* (1995), for instance, found that the application for 24 hours of the herbicide Velpar L (hexazinone) resulted in a fleeting (one-day) reduction in chlorophyll a in the periphyton but no apparent effects on the invertebrates of experimental stream channels.

11.3.2 Toxicity in the field

Effects of toxins at population and community level have been termed "secondary effects" by Hurlbert (1975). Secondary effects are defined as "changes that take place in an ecosystem following and as a result of direct biocide (primary) effects on the growth, survival and reproduction of sensitive species". These effects, which include decreasing interspecific competition, replacement of susceptible by tolerant species, evolution of genetic resistance, migration, and recolonization, are dependent on the nature and 'target' of the biocide. Among the most commonly reported secondary effects are (Hurlbert 1975, p139):

- increased abundance of macroscopic filamentous algae following insecticide-caused mortality of herbivorous insects
- reduction of invertebrate populations dependent on aquatic vegetation as a physical substrate or hiding place (biomass decreased after herbicide application)
- increased populations of benthic detritus feeders following the destruction of aquatic vascular plants with herbicide
- increased abundance of certain invertebrates following insecticide or piscicide-induced mortality of predatory invertebrates or fish

- increased abundance of certain invertebrates following reduction of populations of their competitors, and increased abundance of predators that can utilize the replacing species
- increased growth rates for surviving individuals in food-limited populations whose densities are lowered by biocide-induced mortality
- population declines of species that cannot sustain the competition of other species which have increased following biocide-induced mortality of a predator that preyed preferentially on these other species.

Field-based studies are not nearly as common as laboratory- or mesocosm-based studies but those that do exist have been able to show community-wide effects. For instance, Raven and George (1989) followed the decline and recovery of the invertebrate assemblage in an English stream after an accidental spillage of chlorpyrifos. These authors noted that, as expected, all aquatic arthropods were killed, while molluscs and annelids survived. The sediments remained contaminated for months after the insecticide could no longer be detected in the water itself. While most taxa had recovered within 18 months of the accident, a species of beetle and one of caenid mayfly had not returned after two years.

Davies & Cook (1993) recorded catastrophic invertebrate drift and transient effects on trout in a Tasmanian stream immediately after cypermethrin (a pyrethroid insecticide) was sprayed onto a eucalypt plantation through which it flowed. Schulz (1998) analysed the effects of parathion- and fenvalerate-contaminated surface runoff on the invertebrate assemblage of a headwater stream in northern Germany and showed that numbers of virtually all of the invertebrate species he examined were reduced, in some cases for several months after the runoff events. In contrast, Maloney (1995) found no significant differences in invertebrate numbers in a stream in New Zealand that has been contaminated by the herbicide triclopyr in an attempt to eradicate alien plants.

11.3.3 Synergism and antagonism

One might expect that the combined effects on an organism of any two toxins would be additive - in other words that each would act independently of the other. In fact this is often not so. Some pesticides and other toxins have a greater-than-additive effect and are said to act *synergistically* to one another, while others have a less-than-additive effect and are said to act *antagonistically* to one another. From a practical point of view this means that one cannot easily predict the overall effects on aquatic ecosystems of pesticides applied sequentially over the season, or appearing in rivers as runoff after rains.

11.4 BIOACCUMULATION

Bioaccumulation is the result of the ability of a living organism to concentrate and accumulate, and thus "magnify" in its tissues, a chemical substance, either directly from the surrounding medium or indirectly through the food-chain (Bruggeman 1982 cited by Jaffe 1991). Accumulation directly from the surrounding medium is called *bioconcentration* and is defined (Bruggeman 1982 cited by Jaffe 1991) as 'the phenomenon whereby a chemical substance accumulates in an organism by direct contact with the surrounding medium through oral, percutaneous, or sometimes respiratory courses'. Barron (1990) reviews the topic of bioconcentration of water-borne organic substances. *Biomagnification* is the 'phenomenon whereby a chemical substance accumulates in an organism through different trophic levels in the food chain' (Bruggeman 1982 cited by Jaffe 1991).

Many biocides are fat-(lipid)-soluble and these usually become stored in the lipid fraction of organisms consuming them. It is this accumulation that is particularly important to humans, who are often positioned at the top of the trophic pyramid, consuming animals that have already accumulated biocides in their tissues. The use of biocides to enhance food production may, if sufficient control and training in their application is not ensured, lead to contamination of protein sources (fish and shellfish), which are particularly important in developing countries (e.g. Lemke 1983). A classic example of biomagnification of a persistent biocide, DDT, up the food chain is presented in Table 11.1 (from Woodwell *et al.* 1967 cited by Odum 1971, but see also Schmitt *et al.* 1983; Stendall *et al.* 1988; Ohlendorf *et al.* 1988). Data on quantities of pesticides and other contaminants in various South African organisms are available for the Crocodile River (Roux *et al.* 1994) and the Olifants River (Grobler 1994), and for fish in various rivers (Heath & Claassen 1999).

Table 11.1 An example of concentration up the food chain of a persistent biocide, DDT, and its residues (mg l^{-1}) DDT + DDD + DDE, on a wet-weight, whole-organism basis. (From Woodwell *et al.* 1967 cited by Odum 1971).

	Total residues mg l^{-1}
Water	0.00005
Plankton	0.04
Silverside Minnow	0.23
Sheephead Minnow	0.94
Pickerel (predatory fish)	1.33
Needlefish (predatory fish)	2.07
Heron (feeds on small animals)	3.57
Tern (feeds on small animals)	3.91
Herring Gull (scavenger)	6.00
Fish hawk (Osprey) egg	13.80
Merganser (fish-eating duck)	22.80
Cormorant (feeds on larger fish)	26.40

Although it is possible to measure biocide residues in the fatty tissue of organisms, the significance of these levels for the organisms has not been established (Addison 1976). It is not clear what a particular concentration in the fat deposit of a particular organism means in terms of its physiological or biochemical well-being. It is clear, though, that when animals are stressed or starved, fat is broken down and fat-soluble residues are released into the tissues, thus imposing an extra stress.

11.5 GENETIC RESISTANCE TO PESTICIDES

Resistance to pesticides by pest organisms is becoming a major problem worldwide. It is perhaps best known that many bacteria have become resistant to antibiotics (the bacillus that causes TB is particularly resistant to virtually all antibiotics), and that the parasite *Plasmodium*, which causes malaria, has become resistant to various anti-malarial agents. The same biological phenomenon has occurred with regard to insects becoming resistant to pesticides. The mechanism of action is well known (see any textbook on evolution for details). Essentially, mutant forms of insects arise by chance in each generation, whether insecticides happen to be present in the environment or not. If it so happens that one of these mutations gives its possessor a selective advantage over others in the population (in this case, not dying in the presence of the insecticide), then it will survive and pass the mutated gene on to its offspring, who in turn will be able to survive and pass the same gene on to their offspring. Pretty soon, we have a large population of resistant individuals. For this reason, we need to be extremely circumspect in the frequency that we use insecticides because although the insecticide does *not* 'cause' the mutation to occur, it *does* provide a selective advantage to individuals carrying the mutated gene. It is this process that has resulted in insecticide resistance in many species of mosquito in many parts of southern Africa; such resistance has meant that the far more toxic and persistent DDT and other organochlorine insecticides are again being used in southern Africa. It is not clear how soon the mosquitoes will develop mutants that protect them against DDT, or how soon the genes that protect organophosphorus-resistant mutants will disappear from the population now that these chemicals are no longer effective, and therefore no longer used.

11.6 THE NATURE, MODES OF ACTION AND TOXICITY OF BIOCIDES

Biocides include acaricides, algicides, anthelminthics, avicides, defoliants, fumigants, fungicides, herbicides, insecticides, molluscicides, nematocides, ovicides, piscicides and rodenticides (Benson 1969 cited by Livingston 1977, Genis & Rabie 1983 cited by Barlin-Brinck 1991). Many of these vary in their mode of toxic action, environmental persistence and pathways of decomposition. This review focuses on those biocides most commonly used and most potentially dangerous to aquatic biotas. PCBs (polychlorinated biphenyls), which are largely associated with industrial effluents, are not included, although they are similar to organophosphorous insecticides (Livingston 1977) in their potential impact on aquatic environments.

11.6.1 Fungicides and herbicides

Herbicides are substances used to control plant growth and are therefore obviously toxic to plants. Some are general metabolic poisons and may therefore be toxic to animals and micro-organisms too. Both fungicides and herbicides include a broad spectrum of organic and inorganic compounds with various toxicity ranges, levels of chemical stability and environmental persistence. This makes generalizations difficult, although relative to certain insecticides they do not persist in the aquatic environment and do not biomagnify to any degree in aquatic food chains (Livingston 1977). Herbicides are usually divided into five categories, depending on their effect on plants. These are:

- Inhibitors of photosynthesis (e.g. triazines, uracils)
- Photosynthetic energy deviators (e.g. paraquat)
- Inhibitors of chlorophyll formation (e.g. amitrols, pyrichlor)
- Uncouplers of respiration (e.g. 2,4-dinitrophenol - DNP, dinoseb)
- Growth regulators (e.g. 2,4-D, 2,4,5-T)

11.6.2 Insecticides

Insecticides include some of the most frequently used biocides. Hellawell (1986) compared the main characteristics of synthetic insecticides as follows (Table 11.2):

Table 11.2 Comparison of the main characteristics of synthetic insecticides (from Hellawell 1986).

Characteristic	Organochlorines	Organophosphates	Carbamates
Potential for entry into fresh water	strong	strong	moderate
Solubility in water	very low	low	low
Aquatic toxicity	high	moderate	moderate
Aquatic persistence	prolonged	short	short
Bioaccumulation potential	strong	weak	weak

11.6.2.1 Organochlorines

This group includes the chlorinated hydrocarbons (e.g. DDT) and the cyclodienes (e.g. aldrin, dieldrin, endrin). They are inexpensive to produce and very effective in pest control, making them cost-effective. They are however very persistent in the environment, largely insoluble in water, photostable, highly toxic to many organisms (Livingston 1977) and have a tendency to biomagnify and bioconcentrate through food-chains. The organochlorine insecticides interfere with calcium metabolism and affect the nervous system by moving into the nerve membrane. The infamous DDT (and its derivatives DDD and DDE), dieldrin and mirex are in this group. The US Fish and Wildlife Service has produced useful reviews of the environmental hazards of a number of organochlorine compounds including toxaphene (Eisler & Jacknow 1985), mirex (Eisler 1985a) and pentachlorophenol (Eisler 1989a).

11.6.2.2 Organophosphates

Organophosphate insecticides, which are now very widely used, are much more biodegradable and less subject to biomagnification than are the organochlorines, and are also usually unstable in the presence of sunlight. Their toxic action is by the inhibition of acetylcholinesterase, an enzyme involved in the transmission of nerve impulses in virtually all animals. A major problem with any insecticide having such a universal effect on animals is that non-target species are often even more affected than is the targeted pest species. (In the case of organophosphorus insecticides, tests for cholinesterase inhibition are useful for estimating the degree of incidental contamination of non-target organisms.) Furthermore, the development of resistance by the target organisms often forces the user to increase doses in order to maintain the required degree of effectiveness of the insecticide. The US Fish and Wildlife Service has produced useful reviews of the environmental hazards of a number of organophosphorus compounds including chlorpyrifos (Odenkirchen & Eisler 1988) and diazinon (Eisler 1986a).

11.6.2.3 Carbamates

Carbamates share many properties, including inhibition of acetylcholine esterase, with the organophosphate insecticides. Eisler (1985b) reviews the environmental hazards of carbofuran, a carbamate insecticide.

11.6.2.4 Pyrethroids

Pyrethroids are analogues of the naturally occurring insecticide pyrethrum, which has been in use as an insecticide for some time. Natural pyrethrin derivatives tend to be less toxic to mammals than most other insecticides are, and they are readily biodegradable. Synthetic pyrethroids prepared in recent years are highly toxic to fish and some experts believe their effects to be similar to that of DDT (Plapp 1981 cited by Barlin-Brinck 1991). Synthetic pyrethroids are photostable and resistant to metabolism, and there is evidence for cross-resistance in insects between DDT and pyrethroids (e.g. Barlin-Brinck 1991).

11.7 ENDOCRINE DISRUPTORS, HORMONE MIMICS OR ENVIRONMENTAL OESTROGENS

In the last few years (e.g.Tyler *et al.* 1995), it has become clear that breakdown products of some biocides, particularly of various insecticides, are having a rather unexpected effect on living organisms: they are mimicking the natural hormones produced in our bodies . The most obvious effects are on sex hormones, resulting in what has sometimes been called the 'feminisation of Nature' (e.g.Cadbury 1997). The topic is a complex one that cannot be dealt with in detail here. For the South African situation with regard to endocrine disruptors, the reader is referred to Meintjies *et al.* (2000), a *companion volume to the present report.*

11.8 PRACTICAL ISSUES

11.8.1 Modes of application and routes of contamination of aquatic ecosystems

To summarise what has been said above, insecticides may occur in natural ecosystems, both in water and in aquatic sediments. These substances are difficult or expensive to detect in small quantities. Furthermore, some also form a variety of breakdown products of different toxicities. They may also interact antagonistically or synergistically with each other, and with other toxins, so that assessing the risk posed by biocide contamination of a particular aquatic ecosystem can be a very difficult, and sometimes an almost impossible, task.

With respect to the potential contamination of aquatic ecosystems with biocides, the method of application (particularly of biocides used for controlling insects and other invertebrate pests) is critical in determining the level of contamination. Aerial application will obviously have a significantly greater effect than will selective application on the ground. Even so, the type of carrier material, and the size and other properties of spray droplets produced, can significantly affect the extent to which non-target surfaces become contaminated. The quantities reaching streams can be significant: for instance, 3% of the biocide triclopyr applied to agricultural land in New Zealand entered the adjacent stream (Wilcock *et al.* 1991).

Furthermore, much of the pesticide that lands up in surface waters (and, indeed, ground waters) was sprayed onto terrestrial crops. Some of the active ingredient may land up directly in the stream as a result of spray drift, while some will be held on the crop or in the soil, and it may then be washed directly by runoff into the stream or penetrate through the soil profile into the ground water (and thence into a stream). Schulz and his group have quantified these effects for the Lourens River in the south-western Cape. While in Europe the major route of contamination seems to be aerial in spray-drift (e.g. Schulz 2001), in the Lourens River the route seems to be mainly *via* runoff after spates, especially after dry spells (e.g. Schulz *et al.* 2001a, b, c).

11.8.2 Reducing environmental impacts

In Europe, much of the effort towards reducing the effects of pesticides on non-target ecosystems has been directed towards methods of application (e.g. Schulz 2001), for instance by controlling the sizes of spray nozzles. Reducing these effects in a very windy area like the south-western Cape is not very effective, though, and other ways are being sought for reducing the amount of active ingredient reaching aquatic ecosystems. Schulz & Peall (2001) and Schulz *et al.* (2001c) have shown, for instance, that a small constructed wetland acting as a detention pond is very effective in reducing loads of pesticides reaching the Lourens River *via* small tributaries draining the surrounding orchards. Dabrowski *et al.* (2002) report on attempts to model, and thus predict, the quantities of pesticides reaching the same river, and show that riparian strips of intact vegetation are very effective in reducing contamination of the streams they line.

11.8.3 Chemical analyses

Quantification of pesticides and their residues is a complex and expensive business and South Africa has relatively few laboratories with the equipment and expertise to conduct the analyses. Quality of biocide analysis is encouraged by nationwide standards of quality control. Although inter-laboratory calibration exercises (ICE) are conducted to ensure analytical quality (Barlin-Brinck 1991), interpretation of the results is often difficult due to variability between samples.

It seems that the complexity and expense of analysis, as well as difficulties in interpretation, that have prevented South African authorities from initiating a proper survey of the extent of contamination of our aquatic ecosystems and groundwater resources. Barlin-Brinck attempted to collate existing information in 1991, but was unable to obtain the cooperation needed. From the data contained in Meintjies (2000) for evidence of endocrine disruptors in the natural environment, however, such a survey is long overdue.

11.9 CURRENT STANDARDS IN SOUTH AFRICA

Giliomee & Glavovic (1992), Rother & London (1998) and Glazewski (2002) discuss the effects of biocides in terms of environmental management in South Africa. Thus far, the South African Water Quality Guidelines for Aquatic Ecosystems (Dept of Water Affairs and Forestry 1996) deal only with atrazine (a herbicide) and endosulfan (an insecticide). Table 11.3 presents data extracted from the most up-to-date (but nevertheless rather old) South African quality criteria for the protection of aquatic life for various biocides (Kempster *et al.* 1980). The interim DWAF water quality guidelines (DWAF 1996) give Target Water Quality Ranges, Acute Effect Value and Chronic Effect Value for only for endosulphan. More detailed toxicity data are available internationally via the USEPA ECOTOX database: http://www.EPA.Gov/ecotox.

Given the criteria published, and the toxicities reported in Table 11.4, it seems probable that, with respect to these biocides, aquatic biotas are afforded sufficient protection. As synthetic biocide production continues, the importance of establishing, prior to its legal release, the potential effect of the biocide on aquatic ecosystems should be obvious.

Since the use of biocides is unlikely to terminate, and thus their potential to enter aquatic systems remains a real risk, it is necessary to be vigilant in the use, application and monitoring of biocides in the environment.

Table 11.3. Water quality criteria for the protection of aquatic life. Concentrations are in $\mu g\ l^{-1}$. SA = South Africa (from Kempster *et al.* 1980). TWQR: Target Water Quality Range, AEV: Acute Effect Value and CEV: Chronic Effect Value from DWAF (1996).

Substance	SA	TWQR	CEV	AEV
Aldrin	0.01			
chlordane	0.025			
DDT	0.0015			
Dieldrin	0.005			
Endosulfan	0.003	< 0.01	0.02	0.2
Endrin	0.002			
Heptachlor	0.005			
Lindane	0.015			
Malathion	0.1			
Methyoxychlor	0.02			
Mirex	0.001			
Parathion	0.008			
2,4-dichlorophenol	4.0			

CHAPTER 12: TRACE METALS

SUMMARY

In most natural waters trace metal concentrations are very low, and thus any increase exposes aquatic organisms to levels not previously encountered. Contamination of water bodies with trace metals is therefore of significance and should be carefully monitored and controlled. Sources of trace metals include geological weathering, atmospheric sources, industrial effluents, agricultural runoff and acid mine drainage. A number of chemical and physical factors modify the toxicity and uptake of trace metals. These include the chemical species of the metal, the presence of other metals and organic compounds, the volume of the receiving water, substratum type, dissolved oxygen, temperature, hardness, pH, and salinity. Biological factors (e.g. life history stage, age, sex, tolerance levels), influence an organism's susceptibility to pollutants. The overall ecological consequences of trace metal contamination of aquatic ecosystems is a reduction in species richness and diversity and a change in species composition. The selective elimination of less tolerant species, with the resultant reduction in competition and predation, may result in an increase in the abundance of more tolerant species. The degree of change is related to the concentration of the metal(s) and the type (chronic, acute, constant, intermittent) and timing (in relation to season and thus flow rate) of exposure. Current standards in South Africa available for selected metals are summarized in Table 12.3.

12.1 INTRODUCTION

The terminology used when referring to a metal varies, and different authors often use the terms interchangeably. For the purposes of this review a "metal" is defined as an element which is a good conductor of electricity and whose electrical resistance is directly proportional to the absolute temperature. Trace metals are defined, in geological terms, as those occurring at 1000ppm or less in the earth's crust. By this definition, sodium (Na), potassium (K), magnesium (Mg), calcium (Ca), titanium (Ti), iron (Fe) and aluminium (Al) are not trace metals. Fe and Al are included in this review, however, because they often occur only in trace amounts in water. Trace metals can be divided into two groups: i) those like cobalt (Co), copper (Cu), manganese (Mn), molybdenum (Mo) and zinc (Zn)] that occur naturally in trace amounts in most waters (and most of which are plant nutrients) and ii) those like cadmium (Cd), lead (Pb) and mercury (Hg) that do not usually occur in measurable amounts in natural waters, are potentially toxic in low concentrations, and have become widely distributed as a result of human activities (Golterman *et al.* 1978).

The main sources of trace metals in water bodies are geological weathering, the atmosphere, industrial effluents, agricultural runoff and acid mine drainage (from both direct discharge and leaching from the spoils of operational and abandoned mines). Many trace metals are employed in, and result from, industrial activities (Table 12.2).

The term "heavy metal" refers to those metals with atomic masses greater than that of calcium (40.078) and excludes Na, K, Mg, lithium (Li), Al and beryllium (Be). Table 12.1 differentiates the metals into three basic categories according to their toxicity and availability in natural aquatic systems:

non-critical; toxic but insoluble or very rare; and very toxic and relatively accessible (from Förstner & Wittman 1981). The "abundant" and "heavy" metals are indicated.

The United States Environmental Protection Agency considers only Be and Hg as "hazardous", and barium (Ba), Cd, Cu, Pb, Mn, nickel (Ni), tin (Sn), vanadium (Va) and Zn as "potentially hazardous" (Duffus 1980). For the purposes of this review, when the term "trace metal" is used, it includes both iron and aluminium because both usually occur only in trace quantities in fresh waters.

Table 12.1 Classification of metals according to toxicity and availability (modified from Förstner & Wittmann 1981).

Non-critical	Toxic but insoluble or very rare	Very toxic and relatively accessible
Na, K*, Mg*, Al*, Li*, Ca*, Fe*, Rb, Sr	Ti*, Hf, Zr, W, Nb, Ta, Re, Ga, La, Os, Rh, Ir, Ba	*Be*, Co, Ni, Cu, Zn, Sn, As, Se, Te, Pd, Ag, Cd, Pt, Au, Hg, Tl, Pb, Sb, Bi

Italics: Atomic mass < 40 (i.e. not a "heavy" metal).
*Abundant in the earth's crust (i.e. not defined as trace metals).

Trace metals usually exert their biological effects (either as essential micronutrients or as toxins) by forming stable co-ordinate bonds in proteins where they function as catalysts in redox reactions (e.g. Fe, Cu, Mo) or form part of the active centre of enzymes (e.g. Co, Zn). The trace metals known to be essential in one or more biological system are Mg (in chlorophyll), Mn (in enzymes), Fe (most importantly in haeme-containing respiratory pigments), Co (in vitamin B12), Cu (in enzymes involved in redox reactions), Zn (in enzymes) and Mo (proteins involved in electron transfer and photosynthesis). The particularly toxic trace metals tend to form more stable bonds with the SH group than with anions such as CO_3^{2-}, HCO_3^-, OH^- and Cl^- and thus affect proteins by combining with their thiol groups and thus altering their functioning. In fact a number of these metals, including cadmium, copper and zinc, induce the formation in mammals of metallothioein, a protein that sequesters them (Suzuki *et al.* 1990).

As yet, we have little in the way of a general theory to explain the mechanisms whereby metals exert their toxic effects. A team of environmental engineers in the USA have addressed this issue. Paguin *et al.* (2000) and DiToro *et al.* (2001) provide a theoretical model to explain and predict acute toxicity of metals as ligands of certain physiologically active sites on proteins, while Santore *et al.* (2001) report on a comparison of laboratory and theoretical results for copper toxicity to flathead minnows and *Daphnia*. The theoretical and experimental results are sufficiently similar to suggest that the proposed model might be a very useful way forward in understanding and predicting acute toxicity of metals.

12.2 CHARACTERISTICS AND MODES OF ACTION OF VARIOUS METALS

The selection criteria for including a metal in this review depended on their classification as "hazardous" and "potentially hazardous" by the United States Environmental Protection Agency (cited by Duffus 1980); their availability and relative toxicity in the aquatic environment, according to Förstner & Wittman (1981); the number of potential anthropogenic sources; and finally the amount of published data on their toxicity. Much of the information in this section has been gleaned from the books *Environmental Toxicology* by Duffus (1980) and *Principles of Ecotoxicology* by Butler (1978).

Table 12.2. Trace metals employed in and discharged from major industries (modified from Dean *et al.* 1972; Klein *et al.* 1974, and O'Connor 1974; cited by Förstner & Wittmann 1981).

INDUSTRY	As	Cd	Cr	Cu	Fe	Pb	Mn	Hg	Ni	Se	Ag	Sn	Zn
Alkalis, chlorine, inorg. chemicals		■	■		■	■		■				■	■
Detergents	■	■	■	■	■				■				■
Fertilizers and herbicides	■	■	■	■	■	■	■	■	■				■
Food processing										■	■		■
Glass, cement, asbestos products			■										
Leather and tanning industry			■										
Nonferrous metal-works		■	■	■		■		■					■
Organic and petro-chemicals		■	■		■	■		■				■	■
Petroleum refining		■	■	■	■				■				■
Photographic		■									■		
Pulp/papermills			■	■		■		■	■				■
Steam generation power plants			■										■
Steel works		■	■	■	■	■		■	■	■		■	■
Textile industry			■										
Vehicle and aircraft plating		■	■	■		■		■	■	■	■		

12.2.1 Aluminium

Aluminium is one of the more toxic of the trace metals and is probably not an essential nutrient in any organism. It is one of the elements whose solubility is strongly pH-dependent, and whose toxicity depends on the chemical species involved. At alkaline pH values, aluminium is present as soluble but biologically unavailable hydroxide complexes. At intermediate pH values, it is sparingly soluble and probably occurs as hydroxo- and polyhydroxo- complexes. Under acid conditions, it occurs as the soluble, available and toxic hexahydrate species ($Al^{6+}.H_2O$ or aquo-Al). The mechanism of toxicity at the biochemical level is poorly understood (see chapter 6). Elevated levels of aluminium in human brains have been implicated in Alzheimer's Disease, although the link has recently been called into question. Aluminium is found in soluble form mainly in acid mine drainage and is becoming a variable of considerable concern in natural waters affected by acid rain, both a the major cause of toxicity in acidified waters and as the basis of a low-pH buffering system (see Chapter 6). Although aluminium is also found in waters made naturally acid by organic molecules, it usually adsorbs onto these large organic molecules and is therefore not available in soluble form even at low pH. Chadwick & Whelan (1992) have edited a very useful volume on aluminium in biology and medicine.

12.2.2 Antimony

Antimony occurs in water as Sb^{3-}, Sb^{3+} and Sb^{5+} ions. Compounds of the trivalent ion are an order of magnitude more toxic than those of the pentavalent ions and include stibine (SbH_3) and antimony tri- and pentachlorides and -fluorides.

12.2.3 Arsenic

Some forms of arsenic are extremely toxic. These include the gases arsine and trimethyl arsine. Arsenates, although not particularly toxic, uncouple oxidative phosphorylation by substituting for the phosphoryl group, and thus interfere with energy metabolism. Arsenites are more toxic than arsenates, inhibiting a variety of thiol-dependent enzymes. Various forms of arsenic are lipid-soluble and therefore accumulate in living tissues. Arsenic itself is carcinogenic in humans and may also be a co-carcinogen. Many living organisms become acclimated to high levels of arsenic. For instance, many salt lakes in the high Andes of South America contain arsenic in concentrations up to several tens of milligrams per litre. The algae living in these lakes form the food of large flocks of flamingoes, which consequently have high levels of arsenic in their tissues. Arsenic occurs frequently enough (e.g. as a result of mining activities and because of its use as a herbicide) and in high enough concentrations to be of environmental concern. Eisler (1988a) reviews its effects on aquatic organisms.

12.2.4 Barium

Barium is one of the trace elements classified by the USEPA as a "hazardous candidate" because of its potential as an environmental contaminant. It causes vomiting and diarrhoea, bleeding of the kidneys and affects the central nervous system.

12.2.5 Beryllium

Beryllium is one of the two heavy metals (the other is mercury) classified by the USEPA as "hazardous", in that even the slightest exposure affects human health. Beryllium is toxic in very small quantities and is accumulated in the body, where it damages mucous membranes and is carcinogenic in mammals. It competes with magnesium for enzyme-binding sites and thus inhibits enzymes such as DNA polymerase, thymidine kinase and alkaline phosphatase. It has also been implicated in damaging the immune systems of mammals by means of a Be-binding protein.

12.2.6 Cadmium

Cadmium is one of the trace metals classified by the USEPA as a "hazardous candidate". It is easily absorbed by mammals, in which it is concentrated by binding with a protein called metallothionein. It is known to inhibit bone repair mechanisms, to replace zinc in zinc-containing metalloenzymes, and to be teratogenic, mutagenic and carcinogenic. Eisler (1985c) reviews the effects of cadmium on aquatic organisms.

12.2.7 Chromium

Chromium is one of the least toxic of the trace metals at low concentrations and is in fact essential for fat and carbohydrate metabolism in mammals, forming part of the "glucose tolerance factors". It occurs in several oxidation states (-2 to +6), of which Cr^{5+} is the most toxic. For some reason as yet unexplained, it is accumulated up to 4000 times by some algae. At higher concentrations it is genotoxic (causing chromosomal aberrations), teratogenic (causing foetal abnormalities), mutagenic and carcinogenic in mammals. Eisler (1986b) reviews the effects of chromium on aquatic organisms.

12.2.8 Cobalt

Although cobalt is an essential micronutrient, being an intrinsic component of cobalamin (one of the B vitamins), it is also toxic in fairly small quantities. The insoluble inorganic cobalt compounds (e.g. hydroxides, carbonates and oxides) are carcinogenic when injected into mammals, while the soluble ones (e.g. chlorides, nitrates and sulphates) are toxic, inhibiting some enzymes and stimulating others. Its toxicity and availability in the environment are low and so cobalt is not generally considered to be a particularly ecotoxic trace metal.

12.2.9 Copper

Copper is a micronutrient, forming an essential part of cytochrome oxidase and various other enzymes involved in redox reactions in the cell. Nonetheless it is toxic at low doses (0.5 ppm for some algae) although its toxicity is reduced in the presence of zinc, molybdenum and sulphate. In the mammalian body, copper is carried bound to proteins (perhaps the same metallothionein that binds cadmium) or amino acids. It is known to cause brain damage in mammals. In water, copper is mobile and soluble at low pH; it precipitates in alkaline conditions and is thus not toxic. It may occur at fairly high concentrations in humic waters but since it is bound to organic molecules it is not available to act as a toxin.

12.2.10 Iron

Iron is an essential micronutrient in all organisms, forming part of haeme-containing respiratory pigments (e.g. haemoglobin), catalases, cytochromes and peroxidases. Although it has toxic properties at high concentrations, inhibiting various enzymes, it is not easily absorbed through the gastro-intestinal tract of vertebrates. Iron compounds are easily oxidized and therefore high concentrations of reduced forms can result in oxygen depletion in the environment.

12.2.11 Lead

Lead is a common and toxic trace metal. It tends to accumulate in living tissues, and in vertebrates, to become immobilized in bone, where it does not exhibit toxic effects. When bone becomes remobilized, as in fever, or as a result of cortisone therapy, or during old age, however, the lead may be released and exert it toxic effects. A major effect of lead is the result of its interference with the synthesis of haeme, an essential portion of the haemoglobin molecule. It also affects membrane permeability, displacing calcium at functional sites and inhibiting the opening of "calcium channels" in membranes, as well as inhibiting some enzymes involved in energy metabolism. It has also been implicated in reduced immune responses in mammals. Eisler (1988b) reviews the effects of lead on aquatic organisms.

12.2.12 Manganese

Manganese is an essential micronutrient, at least in the glycosal transferases important in proteoglycan synthesis in vertebrates, and possibly in other enzymes as well. A deficiency in manganese in vertebrates leads to skeletal deformities but high concentrations are toxic, leading to disturbances in various metabolic pathways. An excess of manganese has been implicated in disturbances of the central nervous system by inhibition of dopamine formation. Hoal (2001) provides a review of the toxicity of manganese to aquatic organisms.

12.2.13 Mercury

Mercury is one of the two trace metals (beryllium is the other) classified by the USEPA as "hazardous", in that even slight exposure affects human health. Mercury can occur in both organic and inorganic forms. The non-ionic inorganic forms (e.g. $HgCO_3$ in water, or mercury amalgam in dental fillings) are the least toxic because they are not easily absorbed by the gastro-intestinal tract, at least in vertebrates. The Hg^{2+} ion, on the other hand, is highly toxic, as are many organic forms of mercury. *For instance, while the arylmercurials rapidly break down to inorganic forms in the tissues*, and are therefore not particularly toxic, the alkylmercurials are stable, with long retention times in living tissues. They are fat-soluble and therefore accumulate, particularly in nervous tissue. Their main effects include an increase in abnormal cell division and enzyme inhibition (probably by replacing or combining with thiol groups in enzymes) and they are genotoxic, causing chromosomal aberrations. Selenium is an antagonist to mercury poisoning.

In sediments, inorganic mercury may be interconverted to alkylmercurials by anaerobic bacteria. Micro-organisms appear to be relatively insensitive to mercury; indeed, nitrogen-fixing bacteria require a certain amount of mercury in their immediate environments, although normal soil levels are 0.0005 - 1 mg l^{-1}. Phytoplankton photosynthesis, on the other hand, is normally inhibited at levels as low as 0.001 mg l^{-1} (although a few species of phytoplankton are known to accumulate mercury). Because of its lipid solubility, mercury tends to be accumulated up the food-chain. For instance, predatory fish such as pike may have mercury levels in their bodies up to 3000 times that found in the ambient water. Mercury is widely used in various manufacturing processes and as a seed preservative and it is as a result of these uses that mass mercury poisonings have occurred. The best known effect of concentration "up the food-chain" was in Minimata in Japan, where most members of a fishing community were poisoned by eating mercury-contaminated tuna. (See an account in Davies & Day 1998.)

12.2.14 Molybdenum

Molybdenum is a trace metal essential for nitrogen fixation by bacteria and occurs in some flavin-dependent metalloenzymes such as xanthine oxidase. It is neither accumulated in tissues nor is it toxic, except at high concentrations. The presence of molybdenum has been shown to induce symptoms of copper deficiency in mammals. Eisler (1989b) reviews the effects of molybdenum on aquatic organisms.

12.2.15 Nickel

Although the effect of nickel deficiencies in some mammals has led to the suggestion that it is a micronutrient, this is unproven. It is certainly toxic, even in fairly small quantities, inhibiting cytochrome oxidase and various enzymes in the citric acid cycle. Nickel becomes bound in various proteins, including metallothionein. It has been shown to be carcinogenic in mammals. Nickel ions tend to be soluble at pH values <6.5. Above a pH of 6.7 they mostly form insoluble nickel hydroxides. About half of the nickel present in most fresh waters is in the ionic form and about half in the form of stable organic (often humic) complexes, many of which readily adsorb onto clay particles. Nickel carbonyl ($Ni(CO)_4$) is probably one of the most toxic forms of nickel, being both water- and fat-soluble.

12.2.16 Selenium

Selenium, although a non-metallic element similar to sulphur, has been included in this chapter because it often combines with metals and sometimes acts as one. It is an essential trace element in animals and bacteria, but apparently not in plants. It forms an integral part of the enzyme glutathione peroxidase in animals, while bacteria have several apparently selenium-dependent enzymes. Despite its essential nature, and the fact that selenium deficiencies occur quite frequently, selenium is toxic at relatively low concentrations because it mimics sulphur and sometimes replaces that element in the thiol groups of the amino acids cysteine and methionine. In mammals it is an irritant and causes erosion of the joints. Toxic concentrations can be carcinogenic and genotoxic. Selenium seems to be an antagonist to the toxic effects of cadmium and mercury, forming Se-Hg- and Se-Cd-proteins. Despite its potential as a toxin, selenium occurs in the environment only in nanogram quantities and is therefore seldom of environmental concern (Canton & Van Derveer 1997). Indeed, it may even be deficient in certain ecosystems.

12.2.17 Silver

Silver salts are generally not very soluble and therefore are easily excreted in the faeces of those animals examined (probably all mammals) but are toxic in low doses to freshwater fish. Some silver salts are known to have "germicidal properties" and may therefore affect the microfloras of aquatic ecosystems. Gorsuch & Klaine (1999)is a volume of papers reviewing various aspects of the toxicity pf silver.

12.2.18 Tellurium

Tellurium is not an essential element. Toxic levels reduce the activities of some enzymes and some chemical species are potential carcinogens and mutagens. Since it is known to be toxic to micro-organisms, tellurium may under certain circumstances be of concern as an environmental contaminant.

12.2.19 Thallium

Thallium occurs only sporadically and in small quantities in the natural environment and is not considered to be a significant ecotoxin. In the past it has been used as a depilatory and in treatment of venereal diseases and is now used as a rodenticide. Its toxic effects are a result of its tendency to bind with riboflavin and flavin-derived cofactors and thus to interfere with energy metabolism. It has also been implicated as a teratogen and a mutagen and was infamously used as a murder agent in politically-motivated attacks in South Africa in the 1980s.

12.2.20 Tin

Tin may be an essential trace element, although the data are not convincing. Inorganic forms are generally not toxic except at high concentrations. Organic tin compounds such as trialkyl and triaryl tin are lipid-soluble and accumulate in the central nervous system where, in very high concentrations, they apparently have similar effects to those of arsenic. Tin is not normally considered to be an important or common environmental contaminant.

12.2.21 Uranium

Although uranium is a major hazard because of its radioactivity, chemically it is not particularly toxic, except at high levels.

12.2.22 Vanadium

Vanadium may be an essential micronutrient for some organisms. It is accumulated several thousand-fold by certain tunicates (sea squirts); the biological significance of this accumulation is unknown. High levels inhibit tissue oxidation and may precipitate serum proteins. Vanadium is thought to inhibit the synthesis of cholesterol, phospholipids and other lipids, and amino acids. Vanadium is not normally considered to be an important or common environmental contaminant.

12.2.23 Zinc

Zinc is an essential micronutrient, forming the active site of various metalloenzymes, including DNA and RNA polymerases. It is known to bind to metallothionein in mammals. There are few reported cases of zinc poisoning, even in mammals. Zinc is often associated with cadmium in natural conditions.

12.3 THE TOXICITY AND UPTAKE OF TRACE METALS IN THE ENVIRONMENT

On entering an aquatic ecosystem, a trace metal may remain in solution or it may become adsorbed onto a particle, thus forming a complex which may then become immobilized in the sediments; or it may deleteriously affect and/or become accumulated in aquatic organisms. The susceptibility of aquatic organisms to trace metals is influenced by various physical, chemical and biological factors that either alter the toxic potential of the metal or change the organism's vulnerability to it.

12.3.1 Physical and chemical factors

- The chemical species of the metals [free ions, inorganic ion pairs or complexes (such as $Cu(H_2O)_6^{2+}$, $CuOH^+$) and organic complexes (e.g. HgR_2 where R=alkyl)] affect their toxicity. For example, the toxicity of iron depends on whether it is in the ferrous or ferric state, and in suspension or solution).

- The combined presence of a metal and other substances in water may result in either synergistic or antagonistic interactions. For instance the interaction of cadmium and mercury may result in the reduced toxicity of both metals (e.g. Musibono & Day 1999, 2001), while the multiplication of toxic effects resulting from metal ion combinations (such as nickel and zinc, copper and zinc, and copper and cadmium) may lead to as much as a five-fold increase in the toxicity of each (Haberer & Normann 1971 cited by Förstner & Wittmann 1981).

- Organic compounds may reduce both acute and chronic toxicity of metals by complexing the free metal ions. Naturally large polyphenolic organic acids are even more efficient at complexing with, and hence reducing the toxicity of, metals such as Al and Cu.

- Dilution influences metal toxicity via dilution within the system. The addition of a metal-containing effluent to a large volume of fast-flowing water will obviously have less effect than its addition to a slow-flowing or reduced volume of river water. Flow rate and discharge vary with rainfall, season, groundwater flow, etc., and water will evaporate from wetlands, facts that effluent managers should determine prior to discharging effluents into aquatic ecosystems.

- The type and mobility of sediments influence the accumulation and remobilization of metals. The accumulation and potential remobilization of trace metals would obviously be greater in reaches of rivers with soft bottoms than in reaches on bedrock. Metals may become unavailable by being incorporated into sediments but of course they do not disappear completely and may be remobilized at some future time.

- At higher temperatures an organism's respiratory activity is elevated, as is the availability of trace metals (faster rates of absorption and release), leading to increased toxicity.

- The amount of oxygen that can dissolve in water decreases as temperature increases, while toxicity usually increases with an increase in temperature. Furthermore, changes in the redox potential, usually in conjunction with a decrease in oxygen potential, often result in the dissolution and release of the hydroxides of trace metals such as iron.

- Increasing water hardness decreases metal toxicity by forming insoluble carbonates or by providing precipitating calcium carbonate that acts as a surface for adsorption of metal ions. The fact that separate standards are permitted for metal concentrations in soft and hard waters in the European Economic Community (EEC) guidelines is indicative of the importance of this factor.

- pH plays an important role in the interactions between trace metals and aspects such as carbonate hardness, a variety of organic compounds, and chemical speciation. pH has been shown either to increase (e.g. Shaw & Brown, 1974) or to decrease metal toxicity (e.g. Whitley 1968). Changes in pH can strongly influence the degree of complexation of metal cations and organic molecules. Low pH also results in production of free metal ions rather than hydroxide (OH^-) or carbonate (CO_3^{2-}) forms, which are less soluble and therefore less toxic.

- Salinity modifies metal toxicity: increased salt concentrations allow the alkali and alkaline earth metals to compete with trace metal ions, thus remobilizing trace metals from suspended material and sediments.

12.3.2 Biological factors

Factors determining the response of an organisms to a toxic trace metal include the organism's life history stage, age and sex, the level of starvation and activity, the degree of physical protection (e.g. a shell or burrow), tolerance levels and acclimation. All of these may also affect the susceptibility of organisms to pollutants other than trace metals, of course.

12.4 ECOLOGICAL CHANGES IN WATERS RECEIVING TRACE METAL POLLUTION

Various studies have examined aspects of the effects of trace metals on algae/periphyton, microfauna, macroinvertebrates and fish.

Firstly, a number of studies have involved *in situ* monitoring of observed effects following the introduction (under natural or simulated conditions) of one or more trace metals into functioning aquatic ecosystems. These studies have focused on structural changes in the biota: species richness, diversity and composition (e.g. Occhiogrosso *et al.* 1979; Leland & Kent 1981; Weber & McFarland 1981; Weitzel & Bates 1981; Leland & Carter 1984; Sheehan & Winner 1984; Deniseger *et al.* 1986; Genter *et al.* 1987; Crossey & La Point 1988; Roline 1988; Clements *et al.* 1989; Whiting *et al.* 1994; Clements 1994; Kiffney & Clements 1994a, 1994b); and functional changes in respiration, metabolism, etc (Malyarevskaya & Karasina 1987; Crossey & La Point 1988) of communities exposed to trace metal pollution. Gerhardt (1993) reviews the impacts of cadmium, zinc, iron, lead and copper on stream invertebrates, especially in acid conditions.

Secondly, numerous laboratory-based toxicity tests have been conducted to determine the lethal and sublethal effects of certain trace metals on aquatic organisms (e.g. Clubb *et al.* 1975; Hutchinson & Stokes 1975; Foster 1982; Khangarot & Ray 1987; Vymazal 1987; de Nicola Giudici *et al.* 1988; Migliore & de Nicola Giudici 1990; Vuori 1994). Förstner and Wittmann (1981) have reviewed all aspects in their book *Metal pollution in the aquatic environment*. Clarke (1974) has reviewed the effects of metal mine effluents on aquatic ecosystems in Canada, and Eisler (1985c, 1985d, 1986b, 1988a, 1988b, 1989b) has reviewed the hazards of various metals to fish, wildlife and invertebrates.

12.4.1 Community changes

Numerous studies have shown that trace metal pollution significantly alters the structure and functioning of the exposed community.

12.4.1.1 Species richness and diversity

Weitzel & Bates (1981) assessed the impact of discharge from an electroplating works by evaluating the effect it had on the community structure of periphytic diatoms in the Muskingum River, Ohio. The copper-rich waste decreased species richness and diversity (Shannon-Weaver and Brillouin indexes) of the diatom community collected from artificial substrata. Occhiogrosso *et al.* (1979) measured cadmium and nickel levels in the sediments of Foundry Cove, New York, which are highly contaminated by discharges from a battery-processing plant. The distribution of benthic macroinvertebrates was strongly negatively correlated with metal concentrations. Roline (1988) noted a decrease in macroinvertebrate species diversity below the point of trace metal (Zn, Cu, Fe, Pb, Mn, Cd) input into the upper Arkansas River, Colorado, and differences in the susceptibility of invertebrates to metal concentrations.

12.4.1.2 Species composition

Weber & McFarland (1981) observed significant changes in species composition of the periphyton community of a eutrophic, calcareous stream (Little Miami River, Ohio) continuously dosed with copper (120 $\mu g\ l^{-1}$). Two numerically dominant species were eliminated by copper, and replaced by three previously non-dominant species which, together with an increase in abundance of certain other tolerant species, led to an overall increase in species diversity. This study emphasises the need to have good prior knowledge of the species present and their relative abundance when assessing the effects of pollution.

12.4.1.3 Biotic interactions

Competition: Leland & Carter (1984) experimentally determined the effect of various doses (2.5, 5 and 10 $\mu g\ l^{-1}$) of copper on species competition of periphyton communities in Convict Creek, California. The primary effect was to reduce competition for limited resources, namely attachment sites, with tolerant species out-competing non-tolerant species, leading to a change in competitive ability and hence in species competition. Wellnitz & Sheldon (1995) found that diatom colonisation of a stream in Vermont, USA, was impeded when ferromanganese-depositing bacteria deposited iron oxides on the stream floor. The authors speculate that these deposits displace periphytic algae, which are otherwise superior competitors for space and nutrients.

Predation: Clements *et al.* (1989) experimentally determined the effect of 6 $\mu g\ l^{-1}$ copper on macroinvertebrate predator-prey interactions in New River, Virginia, and found that it resulted in the prey (mayflies, caddisflies, chironomids) becoming more susceptible to predation. They attributed this increased vulnerability to a change in behaviour that seemed to be related to the increased copper dosage. They stress the need for supplementing single-species bioassays with more environmentally realistic procedures that take into account the many modifying factors in, and variability and dynamics of, natural ecosystems.

12.4.2 Functional changes

Functional changes are changes in physiological aspects of an organism such as respiration, production and metabolism. Crossey & La Point (1988) measured functional responses in the form of respiratory and production rates of periphyton in response to trace metals (Zn, Cu, Ni, Cd) in Prickly Pear Creek, Montana. The authors noted that the algal production/respiration (P/R) and assimilation (AR) ratios were significantly lower at the impact point than at the recovery point. (AR was obtained by dividing the net hourly primary production by the amount of chlorophyll *a*). In another experiment, Malyarevskaya & Karasina (1987) showed that various concentrations of lead nitrate [0.1, 1, 10 and 100 mg l^{-1} $Pb_2(NO_3)_2$] affected the metabolism of invertebrates (gammarids, snails and chironomid larvae), with gammarids rapidly increasing oxygen uptake at low lead nitrate levels.

It should be noted that, in *in vivo* experiments of the kind reported above, the amount of toxicant calculated to produce a certain concentration may not equal the concentrations reached in the stream due to differential speciation of the metal. This phenomenon is one of the most difficult to control, and indeed to measure, and may account for some of the variations in the reported effects of metals on aquatic communities.

12.4.3 Factors modifying community and functional changes

12.4.3.1 Tolerance levels and adaptation of affected organisms

Leland & Kent (1981) *determined the species composition of protozoans and rotifers colonizing* submerged vegetation in Convict Creek, California, during continuous, low-level exposure to copper. The tolerance threshold for copper varied with species but was between 2.5 $\mu g\ l^{-1}$ and 7 $\mu g\ l^{-1}$. Nyberg (1974, 1975 cited by Leland & Kent 1981) noted that microfaunal communities varied in their tolerance to copper and suggested that genetic adaptation to elevated copper concentrations may result in "resistant microfaunal stocks". Genter *et al.* (1987) determined the effect of zinc exposure (0.05, 0.5, 1.0 mg l^{-1}) on algal communities in outdoor, flow-through stream mesocosms along the New River, Virginia. *Zinc concentrations of 0.05 mg l^{-1} or more significantly changed species composition*, with tolerant green and blue-green algae becoming dominant. Species continuously exposed to metals may become "adapted" to these metals if natural selection increases their tolerance levels (Foster 1982). The degree of tolerance of the affected flora or fauna determines species composition and the pattern of reappearance of species, both temporally and down the course of a river.

12.4.3.2 Seasonal differences

Deniseger *et al.* (1986), in a study of periphyton community structure above and below a trace metal mining operation in Myra Creek, Vancouver Island, noted strong seasonal differences in the effect of chronic exposure of metals (Cd, Pb, Cu, Zn) on the organisms. In spring, species diversity above and below the mining site did not differ significantly (attributed to the reduced impact of the trace metals due to lower temperatures and increased flow rate in spring), although in summer diversity was considerably lower below the mining operation (Shannon diversity index: above = 1.08, below = 0.14). This illustrates the importance of establishing the effects of natural variations in abiotic and biotic factors in relation to the effects of pollutants on stream organisms.

12.4.3.3 Exposure time

Sheehan & Winner (1984) compared three separate studies on changes in invertebrate communities exposed to copper for different durations. Shayler Run, Ohio, was experimentally dosed with copper (Winner *et al.* 1975); Elam's Run, Ohio, received intermittent effluent from a metal-plating plant (Winner *et al.* 1980); and Little Grizzly Creek, California, was exposed to trace-metal drainage from tunnels of a non-operational copper mine for nearly 40 years (Sheehan 1980). Species richness was strongly correlated with the copper gradient, and species composition changed dramatically as copper concentration increased, with chironomid larvae making up a very large proportion in polluted areas. An important fact to emerge from the 1980 study was the need to consider the duration and type of exposure (regular vs intermittent), which strongly affected faunal composition. Those invertebrates in streams receiving intermittent copper releases appeared to be less affected than those exposed to continuous copper releases.

Regular chronic exposure may not be lethal have long-term effects (affecting growth, decreasing reproductive success, altering behaviour and decreasing disease resistance) (Sprague 1971; de Nicola Giudici *et al.* 1988), which explains why it is important to study both acute and chronic responses of organisms to toxins.

12.4.3.4 Life stage, age and condition of the organism

Early life stages (e.g. larvae) may be more sensitive to pollutants (Cairns & Scheier 1959, Clubb *et al.* 1975) than are the adults. Shazili & Pascoe (1986), however, noted that sensitivity of rainbow trout (*Salmo* sp.) embryos to Cd, Cu and Zn varied with time after fertilization. The extrapolated 48-h LC_{50} values were higher than those reported in the literature for the later growth stages. They attributed this to the protective layer formed by the chorion of early embryos, which prevents free passage of pollutants. Embryos were least sensitive to Cd, less to Zn and most sensitive to Cu. Exposure to Cd of the eggs of *Chironomus riparius* (Diptera, Chironomidae), which had been separated from their gelatinous matrix, reduced hatching success by approximately 60% (Williams *et al.* 1987). De Nicola Giudici *et al.* (1988) showed that 5 $\mu g\ l^{-1}$ copper did not harm females of the isopod *Asellus aquaticus*, but that it affected juvenile growth rates and survival.

Testing toxicants has often been done on adults alone, but it is clear that the effects on more sensitive early life stages may be more informative and therefore more important for setting guidelines and producing criteria. Anderson *et al.* (1980), for instance, working on early life stages of insects, obtained LC_{50} concentrations 1600 times lower than values for insects reported in the literature. They attributed this to the exposure of insects during critical life cycle events, such as embryogenesis, hatching and larval development into the second and third instars.

12.5 RISK ASSESSMENT

The relatively new discipline of ecological risk assessment (ERA) evaluates the potential for adverse environmental effects resulting from exposure to contaminants. General principles of risk assessment with regard to metals and metal-containing compounds are discussed by Chapman & Wang (2000). In order to assess the risk posed by the presence of contaminants, it is neccesary both to ascertain the degree of contamination (or likely contamination) in the field, and also to be able to estimate what degree of risk that poses; laboratory toxicological data are usually used for this second aspect. The difficulty lies in interpreting the two kinds of data and relating them to each other. Clements & Kiffney (1994) discuss ways in which this can be done when assessing impacts of heavy metals.

12.5.1 Biological assessment of trace metal pollution

Contamination of an aquatic ecosystem can be confirmed by examining the water, the sediments and the organisms. Metals in water are exposed to many different chemical reactions and so analysis of this medium gives a transitory view of the metal loading. Soluble metals eventually reach the sediments, where they adsorb onto clay minerals and organic matter, and essentially disappear, while occasional toxic doses of trace metals may be missed by temporally or spatially widely spaced chemical monitoring. The organism, on the other hand, is constantly active in both water and sediment and will have will have been exposed to everything that occurred in its habitat during its lifetime. Thus biological information is often more valuable than periodic physical and chemical monitoring in assessing the long-term effects of pollution.

Many studies have assessed the value of different organisms in monitoring changes induced by the addition of toxic compounds. Good biological indicators are often good accumulators of trace metals. Förstner & Wittmann (1981) give a detailed account of the accumulation of trace metals in marine and freshwater organisms at various trophic levels. Organisms used to measure metal accumulation include algae/periphyton (Keeney *et al.* 1976 cited by Förstner & Wittmann 1981; Ramelou *et al.* 1987), macrophytes (Bonforte *et al.* 1986; Sridhar 1986; Reimar & Duthie 1993; Nelson & Campbell 1995), invertebrates [including aquatic insects (Nehring 1976, Nehring *et al.* 1979) and crustaceans (Plénet 1995)] and fish. Hutchinson & Stokes (1975) reviewed the role of algal bioassays in determining trace metal pollution, and Vymazal (1987) focused on reviewing the toxicity and accumulation of cadmium with respect to algae and cyanobacteria. A somewhat different approach examines morphological deformities apparently resulting from exposure to metals (e.g. Janssens de Bisthoven *et al.* 1992).

Quantitative information about contaminants accumulated in organisms may be useful. Peterson *et al.* (1996), for instance, report on studies on streams in Tennessee, USA, that receive a variety of effluent discharges. The small resident fish accumulate mercury and other heavy metals, the degree of accumulation being proportional to the degree of contamination of the streams in which they live. They can thus be used as biomonitors of contamination. In a case of immediate human interest, contamination of Lake St. Clair, Detroit, by mercury resulted in the prohibition of the sale of fish, which had an average mercury content of 0.3-5 mg kg^{-1}. (According to the World Health Organization, the maximum allowable level in human food is 0.5 mg kg^{-1}).

Table 12.3 Current guidelines developed for selected trace metals in South Africa (DWAF 1996). TWQR: target water quality range; CEV: chronic effect value; AEV: acute effect value. * : concentration dependent on pH and/or water hardness.

Metal (mg l^{-1})	Symbol	TWQR	CEV	AEV
Aluminium	Al	0.005 - 0.001*	0.01 - 0.02*;	0.1 - 0.15*
Arsenic	As	0.01	0.02	0.13
Cadmium	Cd	0.00015 - 0.004*	0.0003 - 0.0008*	0.003 - 0.0013*
Chromium	Cr	0.007 - 0.012*	0.014 - 0.024*	0.2 - 0.34
Copper	Cu	0.0003 - 0.0014*	0.0005 - 0.0028*	0.0016 - 0.012*
Iron	Fe	Not vary by > 10% background concentration		
Lead	Pb	0.0002 - 0.0012*	0.0005 - 0.0024*	0.004 - 0.0016*
Manganese	Mn	0.18	0.37	1.3
Mercury	Hg	0.00004	0.00008	0.0017
Selenium	Se	0.002	0.005	0.03
Zinc	Zn	0.002	0.0036	0.036

12.6 CURRENT STANDARDS IN SOUTH AFRICA

Effluent standards and water quality guidelines are based mostly on ecotoxicological data but of course those data are imperfect; moreover, they seldom cover a wide enough variety of taxonomic groups, nor are they available for combinations of toxic substances. Janssen *et al.* (2000) discuss the uncertainties involved in deciding guideline values. Ecotoxicological values established for various trace metals have been used used in the development of the interim South African Water Quality Guidelines for the protection of natural ecosystems (DWAF 1996). Table 12.3 lists current guidelines or standards for South Africa. More detailed toxicity data are available internationally via the USEPA ECOTOX database: http://www.EPA.Gov/ecotox.

CHAPTER 13: AGRICULTURE

SUMMARY

Agricultural activities may have considerable impacts on aquatic ecosystems. Continually expanding global food production is likely to lead to further impacts unless cognisance is taken of the impacts and measures are taken to mitigate them. Agricultural processes such as land preparation, irrigation, fertiliser application, livestock handling and pesticide application may all negatively impact on receiving water quality. Turbidity and sediment may increase during land clearing, burning and ploughing and under conditions of heavy grazing. Return-flow irrigation water, both surface and subsurface often leads to elevated concentrations of salts and causes salinisation of associated water bodies. Land clearing, fertiliser application and wastes from livestock causes an increase in nitrogen and phosphorus concentrations. Discharge of organic wastes from livestock results in a high BOD of the receiving water body and may be a source of bacteriological contamination. Pesticides and herbicides may enter the stream or river via surface runoff. Several options exist for reducing the impact of agricultural activities on aquatic ecosystems, including the maintenance of a riparian zone buffer strip, appropriate application of fertilizer in terms of quantity and timing, exclusion of livestock from the stream- or riverbanks, storage and treatment of waste products and discharges (e.g. manure, subsurface irrigation water), and vigilant control of pesticide/herbicide application and avoidance of aerial spraying if possible.

13.1 INTRODUCTION

The doubling of agricultural food production globally over the past 35 years have been coupled with a 6.87-fold increase in nitrogen fertilisation, a 3.48-fold increase in phosphorus fertilisation, a 1.68-fold increase in the amount of irrigated cropland, and a 1.1-fold increase in land in cultivation (Tilman 1999). These values were presented in a paper (Tilman 1999) on the global impacts of agricultural expansion, together with predicted future increases associated with further doubling of global food production. Current agricultural practices involve deliberately maintaining ecosystems in a highly simplified, disturbed, and nutrient-rich state. Limiting factors, especially water, mineral nitrogen, and mineral phosphorus, are supplied in excess, and pests are actively controlled (Tilman 1999). The predicted increase, particularly in nitrogen and phosphorus release from agricultural fields, is expected to have dramatic impacts on the diversity, composition and functioning of natural ecosystems of the world, principally through the eutrophication of aquatic ecosystems (Tilman 1999). Not only is there a threat of further eutrophication, past land-use activities, particularly agriculture, are thought to cause long-term modifications to and reductions in aquatic diversity, regardless of subsequent reforestation of riparian zones (Harding *et al.* 1998). Given these predicted and historical consequences of agricultural activities, it is essential to understand the impact of agriculture of aquatic ecosystems.

13.2 EFFECTS OF AGRICULTURAL ACTIVITIES ON WATER QUALITY

Diffuse pollution arising from agricultural is a major source of problems in the aquatic environment (e.g. Cooper 1993). Various processes such as land preparation (land clearing, burning, ploughing), irrigation, fertilizer application, livestock handling and pesticide application may all influence receiving water quality. The potential for agriculture to affect aquatic ecosystems depends on the type and extent of the pasture, crop, stock, etc. Agriculturally derived pollutants include plant nutrients (nitrogen and phosphorus) from fertilisers and animal manure, organic wastes and low levels of pesticides. Either of theses classes of pollutants may be in dissolved form within the runoff water, or be adsorbed onto particulate soil material washed from the land (D'Arcy &Frost 2001). Agricultural runoff may be surface or subsurface. Surface runoff is governed by rainfall characteristics, particularly intensity and duration, by vegetation cover, soil type and slope (McColl & Hughes 1981).

13.2.1 Turbidity and sedimentation

Sediment yields are often high in rivers draining agricultural land (Allan *et al.* 1997). Soil disturbance accompanying land clearing, burning, ploughing and heavy grazing may lead to increased surface runoff thereby causing sediment to enter the stream or river. The amount of sediment lost from a catchment depends on local factors such as slope and soil type and on the presence and extent of the riparian zone buffer strip. Livestock watering at the bank-side may also cause substantial erosion and sediment input into adjacent water bodies.

13.2.2 Salinisation

In arid regions, irrigation of agricultural land is a common practice. Irrigation is often several times the natural precipitation rate (Lemly 1994). Two types of wastewater are produced during the irrigation process: surface runoff and subsurface drainage. Surface runoff occurs because of operational spillage as water is pumped into canals for distribution to fields, or because application rates exceed soil filtration rates (Lemly 1994). Irrigated agriculture has changed the natural hydrologic regime and greatly accelerated the rate of salt deposition in and salinisation of aquatic systems, particularly wetlands (Lemly 1994). The effect is exacerbated by the abstraction of water inflows into water bodies such as wetlands, which would naturally flush away excess salts.

In arid regions such as southwestern Australia sea-salt is transported to inland regions by wind and rain. Replacement of deep-rooted, perennial indigenous vegetation with shallow-rooted, annual agricultural species has altered the water balance of catchments and increased groundwater discharge (Schofield *et al.* 1988 cited by Kay *et al.* 2001). Consequently, groundwater levels have risen and salt previously stored in the soil is mobilised and brought to the valley surface or discharged directly into streams. This process is referred to as dryland salinisation. Agricultural clearing, i.e. clearing of land for agriculture, has also changed the salt balance of catchments from a state of salt equilibrium or accumulation to a state of net salt export (Schofield *et al.* 1988 cited by Kay *et al.* 2001). Stream salinities in southwestern Australia have doubled in the medium-high rainfall zone (> 700 mm/y) and increased up to 50-fold in the low-medium rainfall zone (300 – 700 mm/y) (Scholfield *et al.* 1988 cited by Kay *et al.* 2001). Locally, the Berg River in the Western Cape is highly saline due to irrigation of land and concentration of salts (Fourie & Görgens 1977).

Subsurface drainage is produced due to specific soil conditions (i.e. clay layers) that impede the vertical and lateral movement of water as it percolates downwards. This results in waterlogging of the root zone and subsequent build-up of salts as excess water evaporates from the soil surface (Lemly 1994). To alleviate this, pipes are laid 1-3 m below the surface to facilitate liberal application of irrigation water. The resultant subsurface wastewater is pumped or allowed to drain into surface canals and ditches for discharge into streams or rivers. Subsurface irrigation drainage is a complex effluent characterised by alkaline pH, elevated concentrations of salts, trace elements, and nitrogenous compounds, and low concentrations of pesticides (Lemly 1993). The immediate impact of subsurface irrigation water is the degradation of surface- and groundwater quality through salinisation or potentially toxic trace elements (e.g. arsenic, boron, chromium, molybdenum, selenium) (Lemly 1994).

13.2.3 Nutrients

The clearing of land for agriculture can result in the release of significant quantities of stored nutrients, by altering water and nutrient runoff patterns and increasing soil erosion. Conversion of land for agricultural purposes usually requires application of fertilizers (McColl & Hughes 1981). Both land clearing and application of fertilisers may increase nitrogen (N) and phosphorus (P) concentrations in water bodies associated with agricultural activities. Wastes from livestock are also rich in N and P and unless properly recycled into arable lands, or subjected to tertiary sewage treatment to remove N and P, such wastes can be a major source of N and P loading to aquatic ecosystems (Tilman 1999). The effects of nutrient enrichment on aquatic ecosystems are discussed in chapter 10.

13.2.4 Organic enrichment/dissolved oxygen

Organic waste discharged from livestock farms or deposited into stream during livestock grazing, are considered to be oxygen-demanding. When discharged into a receiving water body micro-organisms (bacteria, fungi, protozoans) use up dissolved oxygen while consuming or decomposing the waste. The rate at which this process occurs is measured as the *biological oxygen demand* (BOD). Manure may have BOD concentrations in excess of 1000 mg O-1 (Bloxham 1999). The importance of dissolved oxygen in aquatic ecosystems in discussed in chapter 8, whilst the effects of organic enrichment are detailed in chapter 9.

13.2.5 Bacterial contamination

Livestock waste is a source of bacteriological contamination (Cooper & Lipe 1992, Brenner & Mondok 1995). It may occur when livestock manure is deposited in the stream either directly or during surface runoff. Fecal coliforms (FC) and fecal streptococci (FS), whilst not pathogenic, are the primary indicators of the potential presence of pathogens (Larsen *et al.* 1994). Pathogenic organisms, when present in animal waste, can be transferred to humans via water (Diesch 1970 cited by Larsen *et al.* 1994). Some potential diseases that may be transferred to humans via cattle, for example, are salmonellosis, anthrax, turberculosis, tetanus, colibacilosus, etc. (Azevedo & Stout 1978 cited by Larsen *et al.* 1994). Peak fecal coliform concentrations are frequently related to runoff events.

13.2.6 Biocides

Biocides are used to control pests associated with agricultural crops and vector-borne diseases. The most commonly used pesticides are herbicides, fungicides and insecticides. Effects on aquatic ecosystems are dependent on the quantity and type of biocide applied and on the method of application. If aerial spraying is done, surface runoff may contain high concentrations of pesticides and herbicides (Lemly 1994). With long-term use, biocides are more likely to be present in low but consistent levels (Cooper 1991) and may pose unknown, sublethal problems. Chronic effects from low levels of herbicides are difficult to detect (Cooper 1993). Some herbicides reduce primary productivity by being toxic to phytoplankton. Others produce indirect effects when oxygen is depleted due to the decay of dead plants. When aquatic herbicides are used to destroy aquatic macrophyte communities, the change in habitat may lead to changes in aquatic community structure (Cooper 1993). Further details are provided in chapter 11.

13.3 EFFECTS OF AGRICULTURAL ACTIVITIES ON AQUATIC BIOTA

Intensive agricultural is known to reduce the richness of macroinvertebrates (Dance & Hynes 1980). Often sensitive species decrease in abundance, whilst tolerant ones increase (Neumann & Dudgeon 2002). Processes associated with agricultural development have resulted in a simplified aquatic ecosystem that has a homogeneous and depauperate macroinvertebrate community consisting of families that tolerate a broad range of environmental conditions (southwestern Australia: Kay *et al.* 2001). Agricultural activities, particularly livestock grazing has also led to changes in aquatic vertebrate communities. Frogs are considered to be very sensitive indicators of environmental change and frog communities, species richness and some individual species of frogs have declined as grazing pressure has increased in the floodplains of the Murray-Darling basin of southeastern Australia (Jansen & Healey 2003). Grazing reduces the quality and availability of aquatic vegetation and thus frog habitat.

13.4 MANAGEMENT OPTIONS FOR REDUCING THE IMPACT OF AGRICULTURAL ACTIVITIES ON AQUATIC ECOSYSTEMS

Solutions to surface water problems caused by agriculture centre around best management practices and sound land use policy (Cooper 1993). Land use policy should dictate that land be used only within the capacity to support agricultural production without abuse (Cooper 1993). Many of the management practices aimed at reducing sediment input into receiving water bodies also helps to reduce nutrient input. The following management options are aimed at reducing sediment, nutrient, organic and pesticide inputs into water bodies associated with agricultural areas.

- Maintenance of riparian zone buffer strips such as multispecies riparian buffer strip system (MSRBS: Schultz *et al.* 1995) to reduce the risk of surface water contamination via surface runoff. The riparian zone is a three-dimensional assemblage of vegetation and organisms adjacent to flowing water that form the ecotone between the terrestrial and aquatic ecosystems (Lowrance *et al.* 1985).
- Avoidance of soil tillage for seedbed preparation and planting during periods of heavy rainfall.
- Exclusion of livestock from stream edges and damp hill slopes, avoidance of overgrazing and exposure of bare soil to reduce the potential for soil erosion and encourage preservation of riparian vegetation.
- Exclusion of livestock from streams and stream-banks to prevent waste entering water.
- Control of grazing pressure to ensure the maintenance of rapid filtration rates (by reducing the amount of trampling), good pasture recovery rates and preservation of the litter layer.
- Controlled runoff from feedlots and proper storage (with impervious linings: Brenner & Mondok 1995) and disposal of manure (manure is suited for application to cropland).
- Application rates of inorganic nitrogen fertiliser that match crop requirements, taking into account the soil nutrient status and any organic manures applied (D'Arcy & Frost 2001).
- Treatment of subsurface irrigation water to remove salts and contaminants, as is done with other municipal and industrial wastes (Lemly 1994), before it is discharged into receiving water bodies.
- Application of regulations that apply to municipal and factory effluents to large-scale livestock factories or heavily fertilised fields (Tilman 1999).

CHAPTER 14: FORESTRY

SUMMARY

Afforestation often occurs in upper catchments that are most sensitive to disturbance. Forestry affects water quality and quantity, specifically low flows, in receiving water bodies. The major water quality concerns associated with forestry are increased turbidity and sediment, increased concentrations of nutrients, changes in light availability and water temperature, changes in energy inputs with an increase in primary production, and potential acidification of surface waters. Forestry may also affect the flow regime of associated water bodies. Aquatic organisms may be affected by forestry activities, with studies reporting effects on periphyton, macroinvertebrates and fish. Primary production (periphyton) has been shown to increase although studies are not conclusive. Macroinvertebrate assemblages generally exhibit a decrease in abundance and diversity and a change in composition. Fish populations are often reduced. Effects on macroinvertebrates and fish are mostly linked to the effects related to increased turbidity and siltation, which affects food and habitat availability. Several options exist for reducing the effect of forestry operations on aquatic ecosystems. Most options focus on limiting the degree of site disturbance through the adoption of minimally destructive practices and the maintenance of riparian zone buffer strips. Buffer strips have been shown to be an effective measure for limiting the movement of sediment and nutrients in the water body whilst limiting modifications to light availability and water temperature. A minimum width (30 m) has been recommended although this may be modified depending on local catchment conditions and type of vegetation.

14.1 INTRODUCTION

Afforestation often occurs in upland areas of catchments that generally have a high conservation status, are sensitive to disturbance and are a critical source of fresh water. In South Africa, *Pinus* spp. and *Eucalyptus* spp. are the primary trees in the forestry industry. Both of these are not indigenous to the region. Few studies have examined the effects of forestry on aquatic ecosystems or, in fact, general biodiversity in South Africa.

14.2 EFFECTS OF FORESTRY ACTIVITIES ON WATER QUALITY

The major water quality concerns associated with afforestation and forestry operations are recognised as being: 1) increased turbidity and sedimentation due to soil disturbance accompanying cultivation, drainage, road construction and harvesting operations (Nisbet 2001); 2) increased concentrations of nutrients, particularly nitrate, ammonium and phosphorus, following fertilisation of intensively managed plantation forests (Binkley *et al.* 1999); 3) changes in light availability and water temperature (Campbell & Doeg 1989); 4) changes in the balance between allochthonous and autochthonous inputs; and possible enhanced capture of acid deposition by forest canopies resulting in further acidification of surface waters (Nisbet 2001).

14.2.1 Turbidity and sedimentation

Soil disturbance accompanying forest ploughing, drainage, road works and harvesting operations has the potential to cause large quantities of sediment to enter the stream or river, resulting in increased turbidity and siltation (Nisbet 2001). Preparation of land for afforestation is normally achieved by land clearing with or without vegetation burning. This, together with road construction for logging, leads to increased suspended solids entering receiving water bodies. The amount of sediment lost from a catchment depends on site factors such as slope and soil type and also on the intensity of the harvesting operation (Campbell & Doeg 1989). Channel form may also be altered as a result of both increased sediment input and modified flow regimes (see section 14.3).

14.2.2 Nutrients

Catchments that have been disturbed by human activity tend to "leak" nutrients (Campbell & Doeg 1989). Soil erosion tends selectively to remove organic constituents and the finer soil particles, which usually have the highest nutrient content (McColl & Hughes 1981). In addition, the application of fertiliser in intensively managed plantations with fertilizers such as urea, ammonium nitrate or diammonium phosphate (Binkley *et al.* 1999), may result in elevated concentrations of nitrate, ammonium and phosphorus in receiving waters. Increased concentrations may, in turn, lead to an increase in primary productivity and potentially to eutrophication of the water body if nutrient enrichment is excessive.

Few studies have assessed the impact of aerial fertilisers on the productivity of aquatic systems (Nisbet 2001). The efficacy of riparian zone buffer strips in moderating the flow of nutrients (and chemicals) into streams and rivers has been indicated (e.g. Lowrance *et al.* 1985, Perrin *et al.* 1984 cited by Binkley *et al.* 1999). A 50-m buffer strip reduced concentrations of urea and ammonium by an order of magnitude and reduced the concentration of nitrate by approximately 60% (Perrin *et al.* 1984 cited by Binkley *et al.* 1999). Forest fertilisation, therefore, commonly leads to moderate increases in streamwater nutrient concentrations and the greatest increases comes when fertiliser is applied directly to streams, when ammonium nitrate forms of fertiliser are used and when the application rates are high or repeated regularly (Binkley *et al.* 1999). Losses to runoff can be greatly reduced by avoiding application when heavy rainfall is forecast and by phasing larger treatments over several years (Nisbet 2001).

14.2.3 Light availability and water temperature

Streams shaded by large stands of riparian vegetation generally have lower summer and higher winter temperatures and a smaller daily range when compared with similar un-shaded streams (Campbell & Doeg 1989). During harvesting it may therefore be expected that both light availability and water temperature would change (Sabater *et al.* 2000). Studies have shown that clear-cutting reduces canopy shading, raises stream temperature and alters stream biology (Newbold *et al.* 1980, Noel *et al.* 1986, Wallace & Gurtz 1986). Clearing next to streams in Brown & Krygier's study (1970 cited by McColl & Hughes 1981) increased temperature sufficiently to interfere with fish life. Changes in temperature are smaller or absent where buffer strips are retained (Brown & Krygier 1970 cited by Campbell & Doeg 1989, Davies & Nelson 1994).

14.2.4 Allochthonous versus autochthonous inputs

Riparian vegetation modifies energy input into streams and rivers in two ways: 1) supplying organic matter and 2) reducing light availability and thermal energy to primary producers. Removal of riparian vegetation reduces inputs of detrital matter and increases light availability. If a forest is clear-cut, the system may shift from an allochthonous to an autochthonous one, with an increase in primary production (Sabater *et al.* 2000) and possible shift in macroinvertebrate community structure as grazers replace shredders. However, this change may only be observed over the long-term (Campbell & Doeg 1989), with a shift from shredders to grazers not always evident in the short-term (Johnson *et al.* 2000).

14.2.5 Acidification

The role of forestry in surface water acidification is controversial (Nisbet 2001). It seems that forestry has a minimal impact on acidification within areas of low acid deposition, however, in areas of high acid deposition both conifer and deciduous forest may contribute to further acidification, principally as a result of the enhanced capture of atmospheric pollutants by forest canopies (DEFC 1991 cited by Nisbet 2001).

14.3 EFFECTS OF FORESTRY ACTIVITIES ON WATER QUANTITY

Afforestation also affects river flows, with flows reduced during reforestation and increased during removal of forest vegetation or harvesting. Trees intercept more precipitation and transpiration rates are greater, resulting in higher evaporation rates compared to other vegetation types (Johnson 1998). Harvesting of trees, particularly clear-felling increases low flows, particularly in the growing season, by reducing interception losses and virtually eliminating transpiration for the first year (Johnson 1998). The effects of forestry on low flows are therefore systematically related to the stage of the forest cycle.

14.4 EFFECTS OF FORESTRY ACTIVITIES ON AQUATIC BIOTA

Long-term biological effects arise mostly as a result of an alteration to the stream riparian vegetation, whilst shorter-term effects are mostly attributable to the impact of suspended or deposited sediment (Campbell & Doeg 1989). No evidence of changes in aquatic ecosystems has been reported from forest fertilisation operations, although few studies have attempted direct examination of the response of aquatic organisms to fertilisation of adjacent forests (Binkley *et al.* 1999).

14.4.1 Periphyton

Data on the effects of forestry, particularly forest harvesting, are inconclusive (Campbell & Doeg 1989) and primary production levels in streams did not have a consistent pattern when streams in logged versus unlogged catchments were compared (Newbold *et al.* 1980). Periphyton was dominated by green algae (Chlorophyceae) in clear-cut streams and by diatoms in the uncut stream (Brown & Krygier 1970 cited by McColl & Hughes 1981). Davies & Nelson (1994) observed an increase in periphyton algal cover at sites affected by logging.

14.4.2 Macroinvertebrates

The major effects on macroinvertebrates are related to elevated turbidity levels and deposition of sediment on the streambed. Changes reported include reductions in species diversity, reductions in biomass and changes in species composition. The effects of increases in turbidity and *sediment are given in chapter 5.* The difference in the total abundance of macroinvertebrates upstream and downstream of logging areas was significantly positively correlated with the width of the riparian zone buffer strip (Davies & Nelson 1994), with Leptophlebiidae (Ephemeroptera) and Plecoptera most significantly affected. Reaches without buffer strips had higher densities of tolerant macroinvertebrate taxa (e.g. Chironomidae, *Baetis*) and lower densities (Davies & Nelson 1994). Kinvig & Samways (2000), in a study undertaken in the KwaZulu-Natal midlands, South Africa, recommended that no commercial trees be planted within 30 m of the stream edge so that light and habitat conditions would favour Odonate species (dragonflies and damselflies). Odonates are considered to be sensitive indicators of water quality and landscape disturbance. Increased algal production and decreased allochthonous inputs may also result in changes in macroinvertebrate abundance and diversity during harvesting, particularly when no buffer strip is maintained (Ulrich *et al.* 2000). Planting of exotic species may affect the food base that many riverine organisms are dependent on. For example, the timing (seasonal versus aseasonal) and nature (nutritional value) of leaf fall may change.

14.4.3 Fish

Forestry activities may have an impact on fish populations indirectly as a result of the impact on macroinvertebrate communities that form the food source of fish, or directly due to habitat modification or lethality. Where adequate strips of intact riparian vegetation are maintained along streams, the impacts on fish populations may be greatly reduced (e.g. Graynoth 1979, Davies & Nelson 1994). A significant positive correlation was found between fish abundance and riparian zone buffer width (Davies & Nelson 1994). The effects of increases in turbidity and sediment are *given in chapter 5.*

14.5 MANAGEMENT OPTIONS FOR REDUCING THE EFFECTS OF FORESTRY ACTIVITIES ON AQUATIC ECOSYSTEMS

Many options are available to the forest manager that will reduce the risk of impacting upon water bodies associated with forested land.

14.5.1 Control of sediment losses to streams and rivers

Options to limit the degree of site disturbance and so reduce sediment losses to streams include:
- adoption of less disruptive practices,
- careful matching of machinery to site conditions,
- varying the timing and scale of operations according to site sensitivity, and
- use of a wide range of protective measures (Nisbet 2001).

Protective measures include:
- limiting the extent of soil disturbance during land preparation,
- the use of protective riparian zone buffer strips along the stream or river (e.g. Newbold *et al.* 1980; Noel *et al.* 1986; Wallace & Gurtz 1986), and
- the siting and design of roads and road crossings to minimise sediment inputs.

The width of the buffer strip varies depending on the channel size and soil erodibility. Davies & Nelson (1994) and Newbold *et al.* (1980) established that buffer strips of $\geq$ 30 m would provide adequate protection for streams in Tasmania, Australia, and in California, United States, respectively. Protective buffer zones do not always provide adequate protection to adjacent streams and rivers as shown in Vuori & Joensuu's study (1996) of moss-dwelling macroinvertebrates associated with *Fontinalis*-tufts in impacted riffles. Sand and silt from forest ditches changed the relative proportion of different habitat types and reduced the quality of the favoured habitat, namely *Fontinalis*-tufts, resulting in an impoverished macroinvertebrate assemblage. Whilst the buffer strip was 30 m, it is described as consisting of sparse vegetation of trees and bushes (Vuori & Joensuu 1996). This highlights the fact that buffer width is not the only factor that determines the efficacy of buffer strips to protect streams from land-use activities. Interception of surface runoff and the degree of impact on receiving waters are also dictated by buffer vegetation characteristics and the overall integrity of the riparian zone buffer strip (Davies & Nelson 1994).

Schultz *et al.* (1995) designed a multi-species riparian buffer strip system (MSRBS) aimed at mitigating the effects agricultural activities on aquatic ecosystems. The MSRBS provides a way to intercept eroding soil, trap and transform non-point source (NPS) pollution, stabilise streambanks, provide wildlife habitat, produce biomass for on-farm use, produce high quality hardwood for the future, and enhance the aesthetics of the agroecosystem. A similar type of buffer system may be worth considering for the control of NPS runoff associated with forestry activities.

14.5.2 Control of nutrient and pesticide input to streams and rivers

The maintenance of riparian zone buffer strips will also moderate the flow of nutrients and pesticides into streams and rivers (e.g. Lowrance *et al.* 1985, Perrin *et al.* 1984 cited by Binkley *et al.* 1999).

The loss of fertilisers to runoff, and thus into the receiving water body, can be greatly reduced by:
- avoiding application when heavy rainfall is forecast,

The effects of aquacultural activities on receiving water quality are related to the discharge of effluents. In particular, elevated concentrations of nutrients from soluble wastes such as urine below effluent outlets and suspended organic material from wastes such as uneaten food and faeces can have a detrimental effect on the aquatic ecosystem with which they are associated (Brown 1992, Brown 1996, Loch *et al.* 1996). The major pollutants in fish farm effluent are considered to be waste food and excreta (Bergheim & Selmer-Olsen 1978), which may settle out on the riverbed. The quantity and quality of solid wastes in the effluents vary seasonally and diurnally depending on the type of feed used, feeding time, stocking rate and other factors. Certain activities such as cleaning of tanks and ponds, and intensive feeding, considerably influence receiving water quality (Bergheim *et al.* 1982). The following section details potential impacts of aquaculture operations.

15.2.1 Nutrient enrichment

Waste food and excreta are rich in nitrogen and phosphorus and increased exchange may take place with sediments. In addition to nutrients associated with the solid fraction, there is a considerable amount of dissolved nitrogen in the effluent in the form of un-ionized ammonia, which is toxic to aquatic life. In fish farms using a simple flow-through system, ammonia will be the principle toxic metabolic by-product, but in recycling systems both ammonia and nitrite may occur at toxic levels in the effluent (Colt & Armstrong 1981). Nitrogen and phosphorus in the effluent (Temporetti & Pedrozo 2000) may lead to hyper-nutrification and stimulation of algal and macrophyte growth in receiving waters. Release rates of phosphorus from the sediments are higher under anaerobic conditions, and calcium and organic matter content of the sediments, rather than the iron concentration, appear to control the availability of phosphorus (Temporetti & Pedrozo 2000).

The extent of the increase in inorganic nutrients in rivers downstream of fish farms probably depends on farm size relative to river discharge, in addition to the original nutrient status of the river (Carr & Goulder 1990b). For example, two of three streams (River Hull and Pickering Beck, England) receiving fish farm effluent experienced marked increases in phosphate concentrations. The absence of change in the third, however, was attributed to high "background" concentrations of phosphate from an upstream sewage effluent. Brown & King (1995) and Brown (1996) noted increases in nitrate, nitrite, ammonium and soluble phosphate concentrations in streams downstream of trout farms.

15.2.2 Organic enrichment and dissolved oxygen

Concentrations of organic matter generally increase below fish farms. Particulate organic material suspended in trout-farm effluent best correlated with changes in macroinvertebrate community structure (Brown 1996) suggesting that this water quality variable has the greatest effect on biotic communities in the receiving water body. Aerobic decomposition of this organic matter by bacteria leads to decreased dissolved oxygen concentrations (increased BOD). Suspended bacteria increased in abundance below the point of enrichment with fish farm effluent in two streams (River Hull and Pickering Beck, England) and this increase reflected more permanent changes in river conditions (Carr & Goulder 1990b). Other factors likely to affect oxygen levels are consumption of oxygen during the breakdown of chemicals (COD) contained in the effluent and indirect downstream effects through changes in phytoplankton abundance. The impacts of changes in the level of dissolved oxygen will depend on the characteristics of the receiving waters and of the effluent, and will certainly have an effect on the survival of natural aquatic biota (Nature Conservancy Council of Scotland 1990). Slow-flowing areas below trout farms in the south-western Cape, for example, exhibited a substantial decrease in the concentration of dissolved oxygen, largely as a result of the settling out of organic material with a concomitant increase in COD and BOD (Brown & King 1995). The settling out of this particulate organic material reduces the quality of the habitat via infilling of interstitial spaces known to be important for aquatic organisms.

15.2.3 Chemical supplements

Various chemicals are used in fish farms to supplement feed, to control diseases caused by various bacteria, fungi, protozoans and viruses, and ectoparasites. These chemicals may enter the aquatic environment in the effluent. They range from fairly benign compounds such as vitamins, to compounds that are extremely toxic to aquatic life. Other anti-microbial compounds include furazolidone, chloramphenicol, quartenary ammonium compounds, sulphonamides, etc (Austin 1985).

15.2.4 Cleaning agents

Cleaning of fish tanks and feeding can cause peaks in the concentration of pollutants in effluents, with peaks in 'cleaning' effluent being 0.1- to 10-fold higher than concentrations of 'normal' effluent (Bergheim *et al.* 1984). These variations have important implications for both the monitoring of fish farm effluents and for the aquatic ecosystems.

15.2.5 Physical structure and suspended solids

Fish-farm operations often lead to an increase in the concentration of suspended solids in the receiving water. The magnitude of the impact depends on the physical structure of the farm (e.g. size, lay-out) and on the type of tanks used. Tanks arranged in parallel may result in a more concentrated final effluent than tanks arranged in parallel. Structurally, there are two types of tanks used in land-based farms in the south-western Cape: unlined earth ponds, and concrete or plastic-lined tanks. Unlined earth ponds have a slower flow-through rate than concrete or plastic-lined tanks and thus some settlement of solids does occur prior to discharge of the effluent into the receiving water body (Brown 1996). The flow of water through tank farms is too fast to allow waste food and excreta to break down before they are discharged into the river. The concentration of suspended organic material in the effluent has been shown to be significantly higher in "portapool type" farms where the flow-through rate of water is much higher, compared to earth dam farms (Brown 1996).

15.3 EFFECTS OF AQUACULTURAL ACITVITIES ON AQUATIC BIOTA

Several water quality variables are likely to increase downstream of fish farms including the concentrations of nitrate, phosphate and ammonium, dissolved solids and suspended solids. It is the combination of changes in these variables that manifests as changes in aquatic biota. The type and location of the fish farm influences the overall impact on aquatic biota. For example, fish farms situated on mountain streams often have a greater impact on macroinvertebrate communities with those in foothill or lowland reaches (Brown & King 1995). This is most likely because lower reaches are often already disturbed by other catchment activities.

15.3.1 Changes in community structure

A study on the effect of trout farm effluents on downstream macroinvertebrate communities in the southwestern Cape, South Africa, has shown that substantial change occurs in community structure (Brown & King 1995, Brown 1996). Of the variables measured, the settlement of suspended organic material and filling up of interstitial spaces, thereby reducing the amount and quality of habitat, was considered important (Brown 1996). The general impact was to eliminate or greatly reduce the number of taxa such as Plecoptera, Limnichidae, Helodidae, Elmidae, Heptageniidae and Telagonodidae (previously called Ephemerellidae), and, in instances where "portapool farms" (use plastic-lined "portapools" for housing fish) were used, to replace these taxa with Naididae, Lumbriculidae, Chironomidae and Planaridae (Brown & King 1995). The upstream community that was dominated by insect taxa was thus replaced by one dominated by non-insects (Brown 1996). EPT (Ephemeroptera: Plecoptera: Trichoptera) taxon richness was also shown to be significantly lower below the effluent outfalls compared to stream sites above their intakes (Loch *et al.* 1996). Recovery of the macroinvertebrate community was noted approximately 1 to 1.5 km downstream of the effluent outlet (Brown 1996, Loch *et al.* 1996).

15.3.2 Toxic effects

Little is known about the potential toxic effects of chemicals and cleaning agents used in fish farms (Nature Conservancy Council of Scotland 1990). Chemicals range from fairly benign compounds such as vitamins, to compounds that are extremely toxic to aquatic life. For example, formaldehyde is toxic to algae at concentrations of 0.3 to 0.5 mg l^{-1}; and malachite green has sublethal effects on fish at concentrations as low as 0.03 to 0.05 mg l^{-1}.

15.3.3 Introduction of invasive exotic species

The likelihood of the escape of invasive exotic species into nearby water bodies needs to be considered, as does the effect of these exotic species on the native biota in the region (Bekker & Brown 1992).

15.3.4 Water abstraction

Whilst the discharge of effluents into receiving water bodies is known to affect aquatic organisms, so too does the abstraction of water from rivers. It can lead to changes in channel shape, patterns of sedimentation, barriers to fish migration and alteration of biological communities (Nature Conservancy Council of Scotland 1990; Jones 1990).

15.4 OPTIONS FOR REDUCING THE IMPACT OF AQUACULTURE OPERATIONS ON AQUATIC ECOSYSTEMS

Several options exist for reducing the impact of aquacultural activities on aquatic ecosystems. Most aim at reducing the quantity of nutrients and organic matter discharged from ponds, especially those receiving manufactured feeds (e.g. Tucker et al. 1996). Options include:

- Designing the aquaculture operation to reduce the concentration of suspended material in the discharge. Specifically, the type of pond, e.g. unlined earth ponds versus plastic-lined tanks, ponds in series versus parallel.
- Decreasing waste production within the pond by reducing the feeding rate, increasing the retention of feed nutrients by fish, or optimising the amount of nitrogen and phosphorus in the feed;
- Increasing the rate of in-pond biological and physico-chemical loss processes for nitrogen, phosphorus and organic matter to reduce concentrations of those substances in the pond before water is discharged;
- Treating the water discharged from ponds using wastewater treatment technologies. Nutrient removal via the use of free water surface (FWS) or subsurface flow (SSF) constructed wetlands has also proved to be effective for the removal of inorganic nitrogen, ammonium, nitrite and phosphate (Lin *et al.* 2002).
- Reducing the volume of water discharged.

CHAPTER 16: ENGINEERING AND CONSTRUCTION

SUMMARY

Engineering and construction include road and highway construction, dam construction and canalisation. All of these engineering activities physically alter the environment and the extent of the modification will determine the resultant effects on aquatic biota. The major physical change in the receiving water bodies is an increase in suspended solids and sediment. The effect on the aquatic ecosystem is partially dependent on the position on the river where the activity is taking place. Generally upper reaches of rivers are more sensitive to physical perturbations than lowland reaches. Other factors affecting the effect on aquatic ecosystems include the duration of construction, the control measures taken and the potential sources of recolonisation of the aquatic biota.

16.1 INTRODUCTION

The main effect associated with engineering and construction relates to the physical disturbance of rivers and their catchments. This may have far-reaching effects on the aquatic biota in the receiving water. The severity of the effect, both short and long term, however, depends on the control measures employed during construction. The effects are complex and depend on temporal (time during and after construction), spatial (distance of the stream reach disturbed), and ecological factors. The area of construction (e.g. an upper headwater stream versus a lowland river) will influence the effects. This relates not only to the potential source of recolonisation of flora and fauna (e.g. invertebrate drift) and to the sensitivity of resident fauna, but also to other pollutional sources which may disguise the effects of the actual construction.

16.2 THE EFFECTS OF DIFFERENT TYPES OF CONSTRUCTION ACTIVITIES

16.2.1 Road and highway construction

The major effect associated with road/highway construction is an increase in suspended solids and deposited sediment (Barton 1977; Chisholm & Downs 1978; Taylor & Roff 1986; Ogbeibu & Victor 1989). Details of the effects of changes in turbidity and suspended solids are discussed in Chapter 5. Barton (1977) examined the short-term effects of highway construction on Hanlon Creek, Ontario, and found that the suspended solid load increased from a mean weekly level of 2.8 mg l^{-1} to 352 mg l^{-1} with a maximum of 1390 mg l^{-1}. The chemical composition of the water did not change during construction. Fish population densities decreased immediately below the construction site although they recovered rapidly after construction was completed. There was no significant change in macroinvertebrate numbers although there was a shift in species composition. Evidence suggested that communities denuded by construction were rapidly replaced by drifting invertebrates recolonising from undisturbed reaches upstream of the site of construction (Barton 1977; Chisholm & Downs 1978). All biotic communities returned to normal within eight months after construction was complete. pH may be temporarily affected during cement pouring if any reaches the water body.

16.2.2 Dam construction

Various studies have shown that earthworks and forest clearing prior to dam construction results in increased suspended and deposited sediments (West *et al.* 1984 cited by Chessman *et al.* 1987; Davey *et al.* 1987 cited by Chessman *et al.* 1987). In Australia, suspended solids increased from a mean of 7.9 mg l^{-1} to 28.4 mg l^{-1} during construction of a dam on the lower reaches of the Tanjil River (Chessman *et al.* 1987). Some of the more sensitive groups, namely elmid (beetle) larvae, plecopterans (stoneflies) and trichopterans (caddisflies) decreased in density. Overall changes in downstream fauna were much less than at some other large dam sites in Australia, probably because of improved erosion control during construction.

In South Africa, all water-resource developments now require an environmental flow assessment (EFA, see Chapter 17, section 17.3) to be undertaken. One of the components of the EFA is that of monitoring. In the case of dam construction, the trend is to conduct both pre- and post-construction monitoring of the downstream environment, as well as monitoring during the actual construction. This will enable more information to be gleaned on the effects of dam construction and allow for mitigatory measures to be developed that further reduce the impact of dam construction of aquatic ecosystems.

16.2.3 Canalisation

One of the primary reasons from an engineer's point of view for canalising a section of river is flood prevention. Threat of inundation of riverside property with water during floods often results when development takes place too close to the river if it does not take into account existing floodplains and anticipated high flood levels. Various problems are often associated with canal construction or channel modification. The key issue is that canalisation cuts off lateral exchange through the riverbanks and vertical exchanges with the hyporheos below the riverbed. Replacement of a heterogeneous bed and banks of a natural river that have a variety of habitats (e.g. stones, sand, backwaters) with smooth concrete has numerous consequences for the biota of the river. Canalisation isolates the river from groundwater input. The loss of natural habitat diversity may affect invertebrate densities and composition as well as fish spawning and rearing. Often protective habitat and feeding sites for adult fish are removed and fish migration is interfered with. Removal of riparian vegetation leads to bank instability, with increased erosion, and increased light penetration with subsequent increases in water temperature. Details of these effects on aquatic ecosystems are discussed in the chapters on temperature (Chapter 4), turbidity and suspended solids (Chapter 5) and urban runoff (Chapter 20).

In South Africa, through projects on rehabilitation of urban rivers (e.g. Davies & Luger 1993, 1994, Luger & Davies 1993), engineers have become more aware of the importance of maintaining the instream-riparian and the instream/hyporheos links. These projects have examined the effects of removing small patches of concrete from the bed of a highly canalised urban river in Cape Town.

The resultant increase in aquatic invertebrates that colonised the new patches indicated that water quality in the river was still reasonable and that removal of stretches of the concrete bed should allow the river to resume some of the ecological processes occurring in a natural river system.

More "river-friendly" building materials, such as Loffelstein blocks, which have been designed, allow the development of some biological communities in the interlocking block, although because of their size and design tend to prevent the growth of riparian trees.

CHAPTER 17: RIVER REGULATION, ENVIRONMENTAL FLOWS AND INTER-BASIN TRANSFERS

SUMMARY

The regulation of rivers in South Africa through the construction of weirs, barrages and dams results from an increasing demand for water by an expanding population. In changing a lotic (flowing) system to a lentic (standing) one, the nature of the water body is altered leading to concomitant changes in river hydrology, water quality and aquatic biotas. River regulation affects river hydrology, through the smoothing of the flow regime. Intrinsic flow variability is often removed and natural floods and fresher floods are dampened or removed. In some instances there is a complete reversal of the natural flow regime. All of these changes may impact on the biota, species of which are often cued into the natural flow regime for spawning periods, and which utilise flooded areas as nursery grounds. Water quality downstream of a dam may be affected; in particular, the values for temperature, dissolved oxygen and nutrient concentrations, turbidity and suspended solids are likely to be different from those upstream. If the water in the reservoir stratifies then downstream effects are somewhat dependent on the depth in the reservoir from which water is released. The effect of river regulation on aquatic ecosystems is currently being addressed within the rapidly evolving field of environmental flow assessment, which attempts to develop methods for establishing the amount of water that needs to be left in a river system, or released into it, for the specific purpose of managing the condition of that ecosystem. The relationship between water quality and water quantity is also been examined, particularly within the context of environmental flow assessment, and a protocol for understanding the relationship between quality and quantity has been developed. Inter-basin water transfers, which are common in South Africa, transfer water from one river catchment to another, or from one river reach to another, have several potential impacts of aquatic ecosystems. The flow regime in the donor and recipient rivers may be modified; species composition may be modified, exotic and invasive species being transferred from system to system; and biodiversity may be reduced through mixing of previously isolated populations. Current legislation in South Africa makes provision for a water quantity "Reserve", which is the amount of water and timing of flow that is required to maintain a given level of ecosystem functioning (National Water Act, No 36 of 1998). The Act also stipulates a water quality component to the Reserve.

17.1 INTRODUCTION

River regulation and inter-basin water transfer occur in response to the need for water by humans. Large natural lentic systems, which act as natural reservoirs, are present in the northern hemisphere. However, in the southern hemisphere, including South Africa, large natural lentic systems are scarce and the bulk of the water supply consists of river water held in storage reservoirs, constructed by humans in the form of barrages, weirs and dams (Davies & Day 1998). Few rivers in South Africa do not have some sort of dam on them and South Africa ranks thirteenth in the world in the number of large dams that have been constructed (Davies & Day

1998). In addition to, and often in association, with weirs or dam, is the transfer of water from one catchment to another. This movement of water, called an inter-basin water transfer (IBT), moves water from rivers with a perceived "surplus" of water to those perceived to be in deficit (Davies & Day 1998). IBTs are an increasingly frequent solution to human demands for water, particularly in arid and semi-arid areas.

The sections in this Chapter on river regulation and IBTs are largely summarised, with the authors' permission, from Davies & Day (1998). River regulation and IBTs have been comparatively well researched in South Africa (e.g. Byren & Davies 1989, Davies 1979, Davies *et al.* 1992, King & Tharme 1994, O'Keeffe *et al.* 1990, O'Keeffe & De Moor 1988). South African scientists are also in the forefront of developing techniques for determining the environmental flow requirements of aquatic ecosystems (e.g. Brown & King 2001, King & Tharme 1994, King & Louw 1998, King *et al.* 1999, King *et al.* 2000, King *et al.* 2002, Malan & Day in press, Tharme & King 1998).

17.2 RIVER REGULATION AND MODIFICATION OF ENVIRONMENTAL FLOWS

Human manipulation of rivers by dams is known as "river regulation". As each dam has its own unique combination of biological, physical, chemical and hydrological characteristics, the effects on the receiving river are variable and complex. Dams form a barrier or a discontinuity in the river. Ward & Stanford (1983) developed a theoretical concept, the Serial Discontinuity Concept (SDC) on the effects of dams on rivers. The SDC proposes that the longitudinal location of a dam on a river, and the depth of the dam from which downstream discharges are made, would determine the extent and the kinds of effects that the dam will have on its river. In addition, each feature that is altered by a dam will probably revert to "normal" conditions (i.e. the conditions that would have pertained were the dam not there) at some point downstream.

17.2.1 Effects of river regulation on river hydrology

The main effect of river regulation is the "smoothing" of the flow regime. Under natural conditions, a river has a flow regime reflective of the rainfall in a region. Different parts of the flow regime elicit different responses from the river ecosystem (King *et al.* 2002). The four kinds of river flow are the low flows, the small floods, the large floods and flow variability, and each different kind of river flow is important in terms of ecosystem functioning (King *et al.* 2002). In South Africa, most rivers are naturally seasonal with summer high and winter low flows in the summer rainfall regions of the centre and east of the country; winter high and summer low flows in the south-western winter rainfall region; and non-seasonal patterns in between (Davies & Day 1998). Some of the modifications to the flow regime that occur when a river is regulated include:

- removal of intrinsic variability and the introduction of flow constancy, either for operational convenience or for specific purposes such as irrigation;
- release of daily pulse flows for optimising hydroelectric power production;
- dampening or elimination of natural floods;
- dampening or removal of freshet floods;
- a complete reversal of the natural flow regime (normal summer base flows elevated above normal

winter base flows) to provide irrigation water in the summer (this is limited to winter rainfall regions), and

- rapid changes in flow rates.

The introduction of unnatural flow regimes may impact on aquatic organisms, including macroinvertebrates and fish. Many fish populations rely on regular floods to recharge pans and inundate river floodplains, which they utilise as nursery areas for juveniles. Under normal flooding conditions, floodwaters recede gradually, allowing juveniles to move slowly back into the river as floodwaters recede (e.g. floodplain below the Pongolapoort dam in Kwazulu-Natal). Factors such as flood heights, intensities and timing all affect the success of this process and post-regulated conditions need to match normal, pre-regulated conditions, to maintain these floodplain fisheries (Coke 1973 cited by Davies & Day 1998).

17.2.2 Effects of river regulation on water quality and aquatic biota

Several water quality variables, including temperature, the concentration of dissolved oxygen, nutrient concentrations, turbidity, the concentration of suspended solids and the concentration of certain chemicals, may be affected in the river below a dam.

The effects are largely caused by the fundamental change that occurs in the water body when it is impounded. As a lotic (flowing) system is changed to a lentic (standing) one, it takes on the characteristics of a lake. In summer this includes thermal stratification. Since warm water is less dense than cold water, it rises to the surface of the standing water body, whilst the deeper water remains cool. This leads to the development of a shear plane, which is evidenced by a thermocline (i.e. a very rapid change in temperature over a very short vertical distance). The phenomenon of separation into a warmer upper layer (the epilimnion), and a cooler, lower layer (the hypolimnion) is known as thermal stratification. The epilimnion is characteristically warm and low in nutrients, whilst the hypolimnion is cool, rich in nutrients and low in dissolved oxygen due to decomposition of material by micro-organisms. In extreme conditions, the decomposition of dead material leads to complete anoxia (exhaustion of the oxygen supply) in the hypolimnion; the chemically reducing conditions consequently lead to the production of large quantities of toxic hydrogen sulphide (H_2S). Most standing water bodies equilibriate in autumn due to the cooling action of the wind and decreasing air temperatures. This process, known as "overturn", results in mixing of the epilimnion and the hypolimnion.

Dams have a release point or points through which discharges are manipulated. The height of the dam wall and position of these release points largely determines the effect on the water quality of the receiving water body.

17.2.2.1 Temperature

During summer, water released from the hypolimnion is often cooler than the river water, whilst water released from the epilimnion is often warmer. hus discharges from the hypolimnion will depress the temperature of the receiving river, while an epilimnetic discharge will elevate it. Most aquatic organisms have a range of temperature at which optimum growth and reproduction occur. Modifying the temperature of the water in which they live is thus likely to affect the aquatic biota of the receiving river. Details of the effects of a change in water temperature on aquatic biota are provided in chapter 4.

17.2.2.2 Dissolved oxygen

Changes in oxygen concentration below a dam are less severe than those described for temperature and flow. This is primarily because oxygen is absorbed by water as it is turbulently discharged from dams. Toxic gases such as hydrogen sulphide may however build up in the dam if the hypolimnion becomes anoxic. During subsequent discharge of hypolimnetic water, these toxic gases may be released. Further details of the importance of dissolved oxygen to aquatic biota are provided in chapter 8.

17.2.2.3 Nutrients

Chemically, the hypolimnion is nutrient-rich in summer during stratification, while the epilimnion is nutrient-poor. The effects on the receiving water body therefore vary depending on the depth of the dam from which the water is released. The effects of nutrient enrichment on aquatic ecosystems are discussed in chapter 10.

17.2.2.4 Turbidity and suspended solids

Unless water is drawn near the bottom of the dam, where sediments may be re-suspended, the water in the dam tends to lose much of its particulate load, and water discharged from dams tends to have less silt than would the natural water. Because water is discharged at a high velocity, water leaving the dam tends to have a "capacity" for silt, and is referred to as being "silt hungry". This state invariably results in rapid erosion of the bed and banks downstream of the dam wall. Erosion of this type is known as bed armouring: all loose fine particles are stripped from the river channel, frequently exposing bedrock and causing the banks to collapse and carry sediments for some distance downstream. Details of the effects of suspended solids on aquatic ecosystems are given in chapter 5.

Summer release of plankton-rich epilimnetic water may affect the receiving river communities since such water is rich in fine particular organic matter (FPOM). Since the quantity and kind of particulate matter varies fairly predictably down a river course, an alteration in the type of particulate matter may significantly alter the types and numbers of organisms in the river downstream of the dam. Organisms known as collectors may proliferate in response to an abundant food source in the form of FPOM. For example, the pest species *Simulium chutteri* (Simuliidae: blackflies), which are functionally collectors of food, became numerically abundant below Vaal-Hartz Diversion Weir. De Moor (1981, cited by Davies & Day 1998) showed that the maintenance of flows during winter when the river would naturally of dried up (this is a summer rainfall region), led to the maintenance of blackfly populations through the winter. This research has led to the use of flow manipulation for controlling pest species, especially ensuring that a dry phase occurs.

17.3 ENVIRONMENTAL FLOWS AND THEIR ASSESSMENT

An increasing number of studies has shown that river regulation and flow manipulations lead to the degradation of aquatic ecosystems. This has lead to the development of the science of environmental flows. Environmental flows may be defined as water that is left in a river system, or released into it, for the specific purpose of managing the condition of that ecosystem (King *et al.* 2002). Briefly, there are four main types of flow-assessment: hydrological, hydraulic rating, habitat rating and holistic (King *et al.* 1999). Of these, the latter or holistic approach formed the basis of the environmental flow assessment (EFA) methods used in South Africa, namely the Building Block Methodology (BBM: King & Louw 1998), and more recently DRIFT (Downstream Response to Imposed Flow Transformations: King *et al.* 2002). Both methods assess the flow requirements at discrete sites that are considered to be representative of the entire river reach in which they are situated, and both methods have an Environmental Flow Assessment workshop as one of the important activities. DRIFT was developed in actual water-resource development projects, on the Palmiet River (Brown *et al.* 2000) and Breede Rivers in South Africa, and in the Lesotho Highlands Waters Project in Lesotho (Metsi Consultants, 2000). DRIFT's basic philosophy is that all major abiotic and biotic components constitute the ecosystem to be managed; and that the full spectrum of flows and their temporal and spatial variability, constitute the flows to be managed (King *et al.* 2002). DRIFT is designed to describe selected potential flow regimes, each linked to the predicted biophysical consequences in terms of condition of the river ecosystem; predicted socio-economic consequences for subsistence users of the river, resulting from the changing river condition; and predicted water yield of the water-resource development (King *et al.* 2002).

DRIFT is a structured process for combining data and knowledge from all the disciplines to produce flow-related scenarios for water managers to consider. In this respect it is a data management tool, allowing data and knowledge to be used to their best advantage in a structured process. DRIFT consists of four modules:

- A biophysical module in which the present nature and functioning of the river ecosystem is described. Predictive capacity of flow-related changes is developed.
- A sociological module is described in which the association between rural subsistence users (including their health) and river resources. Predictive capacity of social impacts of river changes is developed.
- A scenario-development module is developed wherein potential future flows and the biophysical consequences of each flow type are presented as scenarios. The social consequences of each scenario are also described.
- An economic module that calculates mitigation and compensation costs for Population at Risk (PAR) for each scenario.

17.4 LINKING WATER QUANTITY AND WATER QUALITY

Linking the potential effects of altered water quality to the aquatic biota that may result from a change in the flow regime is an essential step in the maintenance of riverine ecological functioning. As part of the environmental flow assessment process, and initially in response to a need for an understanding of changing water quality in response to changes in water quantity, methods for examining the relationship between quality and quantity have been studied (Malan & Day, in press). A protocol using aquatic macroinvertebrates has been developed that comprises three phases, as follows:

- Phase 1 precedes the EFA workshop and involves the examination of the water resource with regard to ecoregion (spatial classification), point-sources of pollution and catchment land-use. Water quality is defined for Reference Condition (RC) and Present Ecological State (PES) for individual river reaches and the appropriate level of protection is established. Flow-concentration (Q-C) plots for each chemical constituent of concern are developed.
- Phase 2 is undertaken at the EFA workshop. Q-C plots are employed to make predictions of water quality in response to the flow regime prescribed by the EFA practitioners. Concentration–exceedence curves are used to compare the water quality consequences arising from the natural, present-day and the recommended flow regimes.
- The third and final phase is undertaken after the EFA workshop. The implications of the predicted water quality for the biota are assessed. Specifically, the likelihood of the Resource Quality Objectives for aquatic macroinvertebrates being attained under the proposed flow regime, and current pollution loading is ascertained.

The application of this protocol should aid in the integration of water quality and quantity, and thereby afford greater protection to aquatic ecosystems under threat from water resource developments.

17.5 INTER-BASIN TRANSFERS

Inter-basin water transfer (IBT) is defined as the transfer of water from one geographically distinct river catchment, or basin, to another; or from one river reach to another (Davies *et al.* 1992). The potential impacts of IBTs are complex, system-specific and depend on initial conditions and river reaches involved. For example, a seasonal river may become perennial one if it receives water via an IBT, whilst the donor river may be affected by the concomitant reduction in flow.

The new National Water Act (Act 36 of 1998) of South Africa gives full power to the Minister of Water Affairs and Forestry to transfer water from river basins with perceived surpluses to basins and human populations with perceived water deficits. Concerns over the impacts of IBTs on river basin integrity, in terms of floral and faunal biodiversity, were first raised (Petitjean & Davies 1988) and pursued by local researchers (Davies *et al.* 1992, Snaddon & Davies 1998, 1999, Snaddon *et al.* 1998, 1999). In particular, research focused on the problems surrounding the genetic integrity of historically isolated river basins and populations and the maintenance of biological processes. Research was continued by Wishart *et al.* (2003) who examined the integrity of catchment units from an evolutionary perspective and assessed the implications of inter-basin water transfers for the conservation of river ecosystem functioning and riverine biodiversity using selected taxa.

The potential impacts of IBTs on aquatic ecosystems include:

- Modification of the natural flow regime, through elevated flows in the recipient river and reduced flows in the donor river;

- conversion of a seasonal river to a perennial one (e.g. O'Keeffe & De Moor 1988);
- transfer of algae (including toxic blue-green algae) from the donor system, particularly if itself is a dam (Snaddon & Davies 1998);
- alteration of the species composition of the recipient system (O'Keefe & De Moor 1988, Wishart *et al.* 2003);
- transfer of pest species and subsequent dominance of these species (e.g *Simulium chutteri*, O'Keefe & De Moor 1988);
- transfer of exotic and invasive species from system to system; and
- reduction of speciation and thus biodiversity through the mixing of previously geographically isolated populations (Wishart *et al.* 2003press).

The most recent research has clearly shown that, with the transfer of individuals from historically isolated populations, the potential exists to undermine the evolutionary processes important in species formation and thus the generation of biodiversity by providing an avenue for gene flow between genetically discrete populations (Wishart *et al.* In press). They urge that all future developments that involve the planning and construction of IBTs should take cognisance of the potential for genetic mixing of riverine biotas in receiving systems.

CHAPTER 18: INDUSTRIAL EFFLUENTS

SUMMARY

Discharge of industrial effluents (treated or untreated) into aquatic ecosystems may affect TDS, TSS, pH, BOD, COD, toxicity (trace metals, toxic organics), colour, nutrients and temperature of receiving waters. The nature of the effluent determines which water quality variables are affected. This chapter describes the constituents of several industrial effluents and indicates which water quality variables are potentially affected by them. Industries examined include the chemical, china-clay, dairy, fertilizer, fish processing, food canning, oil, poultry, pulp and paper manufacture, red meat, sugar, tanning/leather finishing and textile industries.

18.1 INTRODUCTION

The variety of industrial effluents and the potential effects of their constituents on aquatic ecosystems make it unrealistic to discuss them all in this review. Literature examining these potential effects is also relatively scarce, with a few exceptions such as the effects of pulp and paper effluents. The aim of this chapter is to characterise some major industrial effluents in terms of water quality variables affected and the toxins present (Table 18.1).

18.2 INDUSTRIES POTENTIALLY IMPACTING ON AQUATIC ECOSYSTEMS

The following industries have been shown to impact negatively on the receiving water bodies of aquatic ecosystems.

18.2.1 Chemical industry

The large number of commercial chemicals and the diversity of their effects make it difficult to generalise on the industry as a whole (CSIR 1991). Effluents from chemical industries often have high loads of suspended solids and high oxygen demand (as a result of biodegradable organic content and reducing chemicals such as sulphides), in addition to elevated concentrations of trace metals, toxic inorganics (e.g. cyanides, fluorides), organic compounds and nutrients. The pH of receiving waters may also be affected.

18.2.2 China-clay industry

China-clay is comparable to kaolin (hydrated aluminium silicate) and it has been shown to have a similar effect to that of diatomaceous earth (a silicous deposit of diatom frustules: Hellawell 1986). The mining and subsequent manufacture of products in the china-clay industry may result in effluents high in fine, inert solids that have been shown to reduce the abundance of plants and animals in receiving waters (Nuttal & Bielby 1973 cited by Hellawell 1986). Trout survival was significantly reduced in suspensions of china-clay of 270 mg l^{-1} (Herbert & Merkens 1961 cited by Hellawell 1986).

Table 18.1 Water quality variables potentially affected by effluent discharge from various industries. TDS = total dissolved solids, TSS = total suspended solids, B/COD = biological/ chemical oxygen demand, TM = trace metals, TO = toxic organics, C = colour, NUT = nutrients, FOG = fat, oil, grease, TMP = temperature.

Industry	TDS	TSS	B/COD	pH	TM	TO	C	NUT	FOG	TMP
Chemical		■	■	■	■	■		■		
China-clay	■	■	■	■	■	■		■		
Dairy industry	■		■	■			■	■	■	
Fertilizer	■			■	■	■		■		
Fish processing		■	■					■		
Food canning		■	■	■			■			
Oil refinery			■	■	■	■			■	■
Poultry	■	■	■	■					■	
Pulp and paper	■	■	■	■	■	■	■			■
Red meat		■	■	■			■	■	■	
Sugar		■	■	■				■		■
Tanning/leather	■	■	■	■	■	■	■		■	
Textile		■	■	■	■	■	■			■

18.2.3 Dairy industry

A study conducted on behalf of the Water Research Commission in 1989 (NATSURV 4 1989) revealed that the 150 dairies in South Africa use 4.5 million m^3 of water annually. Most major milk production factories are situated near urban centres and their effluent is discharged to municipal sewers, where it is treated prior to discharge into aquatic systems. Most product factories (i.e. those that produce dairy products such as cheese) are in rural areas, however, and discharge of their effluent is more problematic, often causing environmental problems. Much of the effluent derived from milk, butter and cheese production is reused (e.g. as buttermilk) and most wastewater results from associated processes such as cleaning of equipment, bottles, crates, etc.

The majority of effluents in the dairy industry arise from six main areas:

a) wastage of milk and milk products (COD = 75 000 - 86 000 mg l^{-1});
b) plant cleaning-in-process (high COD and elevated concentrations of PO_4 and TKN);
c) crate and bottle washing (high COD);
d) vehicle washing (high COD and elevated concentrations of PO_4 and Total Kjeldahl nitrogen);
e) cheese washing and whey production; and
f) product dumping.

The water quality variables affected are TDS, COD and pH, and the final effluent may have the following concentrations: COD = up to 2000 mg l^{-1}, PO_4 = 12.6 mg l^{-1} (mean), TKN = 9.5 mg l^{-1} (mean). Wastewater may also contain fats, oil and grease. There is great variability in effluent quality between dairies and temporally within dairies (i.e. quality varies on a daily basis).

18.2.4 Fertilizer industry

The production of nitrogenous fertilizers results in wastewaters with relatively high levels of nitrate, ammonia and urea. The production of phosphatic fertilizers results in acidic (low pH) wastewaters high in calcium sulphate, phosphate and heavy metals (CSIR 1991).

18.2.5 Fish processing industry

The processing of fishmeal, oils and canned products results in wastewater with high loads of organics and suspended solids and high concentrations of nitrogen (CSIR 1991).

18.2.6 Food canning industry

This includes the canning of fruit, vegetables and meat. Wastewaters have high concentrations of suspended solids and biodegradable organic loads, high alkalinity, water discolouration and unpleasant odours in the outfall vicinity (CSIR 1991).

18.2.7 Oil refineries

Oil and petroleum refineries produce a wide variety of wastes including oils (free oil or oil emulsion), condensate waters (high COD and possibly ammonia, trace metals, cyanides, phenols etc), acid wastes (acidic, high COD), waste caustics (alkaline), and high loads of organics that may include mercaptans, sulphides, phenols etc), special chemicals (high organic load and toxic substances such as phenols, nitrobenzene, etc.) and cooling water (temperature). Singh & Gaur (1989) studied the extent of oil and phenol contamination in a small stream in the state of Assam, India, receiving refinery effluent. Increased concentrations of oil, phenol and ammonia led to the loss of several algal species, as well as a general decrease in diversity, richness and a change in species composition.

18.2.8 Poultry industry

The 140 white-meat abattoirs in South Africa use 6 million m^3 water annually (NATSURV 9 1989). Wastewater has the following characteristics: pH = 5.9 - 10.1; COD = 689 - 6 780 mg l^{-1}; TSS = 106 - 1 240 mg l^{-1}; TDS = 332 - 2 170 mg l^{-1}. Solid wastes (blood, fat, skin, feathers, viscera, faeces, etc.) are transported and disposed of at solid-waste dumps.

18.2.9 Pulp and paper manufacture

The impacts of pulp and paper mill effluents on aquatic ecosystems are complex and result from interaction of several potentially adverse water characteristics (Davis *et al.* 1988). These include TDS, TSS, BOD, COD, pH, toxicity (potential toxicants include chlorinated phenols, quinones, sulphides, mercaptans, resins, and fatty acids) and colour (Walden 1976 cited by Davis *et al.* 1988). The relative contribution of each to the overall impact varies considerably with pulping process and its efficiency, the species of wood pulped, the waste treatment employed, and the physical and chemical nature of receiving streams (Hutchins 1979 cited by Davis *et al.* 1988). According to a study conducted in 1990 (NATSURV 12, 1990), South Africa has 21 mills, including integrated mills (where pulp- and paper-making are integrated at a single site), non-integrated mills (a single pulp or paper mill) and secondary fibre mills where waste paper is the raw material. Large volumes of water are used (130 million $m^3 a^{-1}$ for 3 million t a^{-1} of paper products) and waste disposal often occurs directly into rivers or the sea, either with or without prior biological treatment (NATSURV 12 1990). The resultant wastewater has a high organic load: COD = 200 - 17 000 mg l^{-1} (NATSURV 12 1990); BOD = 100 - 400 mg l^{-1} (Callely *et al.* 1977 cited by Hellawell 1986) and inorganic load: TDS = 500 - 13 000 mg l^{-1} (NATSURV 12 1990).

The pulping process may be mechanical (without the use of chemicals) or chemical (including soda, sulphite and the current-day Kraft process). The composition of the suspended solids depends on the type of fibre (waste paper or pulp) and filter (clay or chalk) used. Wastewater from the bleaching of Kraft pulp is the major source of pollution at most mills, and may contain degradation products originating from the lignin (may result in surface foaming), carbohydrates, simple phenols, neutral and acidic compounds (NATSURV 12 1990) and chlorine or zinc (CSIR 1991; Hellawell 1986). The organic fractions are generally regarded as non-biodegradable because of the presence of biotoxic constituents such as chlorphenolics.

Available literature on the impact of bleach Kraft mill effluents (BKME) on periphyton and phytoplankton indicates that the effects are, to a certain extent, site-specific. Photosynthetic rates of periphyton exposed to BKME in the Sulphur River, Texas, remained constant, whilst phytoplankton photosynthetic rates were significantly lower downsteam of the input point, probably due a decrease in light. A biological monitoring programme was undertaken above and below the pulp and paper mill discharge from the Australian Paper Manufacturers into the Latrobe River, Victoria (Scarlett & Harris 1990). The discharge formed approximately 4% of the river discharge and biomonitoring included acute bioassays on fish and shrimps, sub-acute bioassays on algae, and *in situ* surveys of the riverine fauna. No acute toxicity to fish or shrimps was noted, algal productivity was not outside the normal range of variation, and benthic invertebrate density and diversity did not change. Changes in species composition were not examined, however.

The effect of a pulp and paper mill effluent on aquatic ecosystems ultimately depends on recovery processes during operation and dilution of effluent. An accidental spill of "black liquor" (high BOD and sulphur-containing substances) resulted in massive fish kills in the Elands and Crocodile rivers in the Transvaal (Kleynhans *et al.* 1992). Primary treatment facilitates the settling out of fibre and suspended solids; aeration ponds remove the BOD material; and dilution and retention in lagoons before discharge (Scarlett *et al.* 1985) all reduce the impact of the effluent on receiving water bodies.

18.2.10 Red meat industry

The 285 red meat abattoirs in South Africa use 5.8 million m^3 of water annually (NATSURV 7 1989). Wastewater quality: pH = 5.7-8.4; COD = 2 380-8 942 mg l^{-1}; TSS = 534-3 300 mg l^{-1}; and TKN = 5-71 mg l^{-1}. Solid wastes were generally disposed of by burning or burying the wastes.

18.2.11 Sugar industry

Water intake by the 16 sugar cane processing plants and one full refinery in South Africa is low relative to that of other industries (NATSURV 11, 1990) because the raw material (sugar cane) has a very high water content (70%), which is used for processing. Additional water is used mostly for cooling and domestic consumption. In South Africa, most plants are in KwaZulu-Natal, with two in Mpumalanga. The primary pollutants in wastewater are enstrained sugar and various process liquors such as juice, syrup and molasses. Much of the solid waste is reused and some wastewater is used for irrigation. The effluent has a high COD (up to 20 000 mg l^{-1}) and biological treatment is hampered by the deficiency of nitrogen and phosphorus. Almost all sugar plant waste is totally soluble, and pollutants are mainly biodegradable organic matter and thermal pollution (CSIR 1991). Recently the use of ash-disposal dams and artificial wetlands has led to great improvements in effluent quality (Table 18.2).

Table 18.2 Improved water quality by use of ash dams and artificial wetlands (from NATSURV 11 1990).

VARIABLE	INFLUENT TO ASH DAM	EFFLUENT FROM ASH DAM
pH	8.5	8.0
COD	1154	71
TSS	2760	155
PO_4	12	10

18.2.12 Tanning and leather finishing industry

There are 20 tanneries in South Africa that produce 2 million hides per year and use 0.6 million m^3 of water per year (NATSURV 10 1989). Numerous processes are involved in the preparation of a hide, including soaking, unhairing, fleshing, deliming and bating, chrome tanning, wet blueing, neutralizing and retanning, dyeing and fatliquoring, samm/setting, vegetable tanning and finishing. Solid wastes include raw-hide and leather trimmings, shavings and buffing dust, fleshings and degraded hair, and sludge settling out from the liquid wastes. In terms of pollution load, the wet blue processing is responsible for the majority of the load discharged by a full tannery. A typical tannery waste may include soil, antiseptics, fats, salt, chrome, vegetable tannins, syntans, dyes and lacquers. The effluent will also be extreme in pH variations, high in TDS, TSS and organic content. Wastewater loads tend to vary greatly over relatively short periods of time. Table 18.3 gives the raw wastewater quality from a tannery.

18.2.13 Textile industry

Textile industries may use natural fibres of animal (wool, hair, silk) or plant origin (cotton, flax, jute, hemp, sisal) and/or synthetic fibres (e.g. rayon, nylon) (CSIR 1991). Pollution characteristics vary from factory to factory depending on the type and quantity of chemicals used, production rates and equipment. Generally wastewater has high acidity/alkalinity, high concentration of suspended solid and biodegradable organic loads, metals (e.g. chromium, mercury), toxic organic compounds (e.g. phenol), colour and warmed cooling waters. The dyes used in the textile industry may also be toxic to aquatic organisms (McKee & Wolf 1963) and are often responsible for the discolouration of the water.

Table 18.3 Raw wastewater quality from a tannery

Variable	Average (mg l^{-1})	Range (mg l^{-1})
pH	8.4	3.9 - 11.1
COD	9700	2760 - 29110
TDS	19600	10100 - 23400
SS	1970	1250 - 4100
Cr	120	42 - 690

CHAPTER 19: MINING

SUMMARY

South Africa has many operational and abandoned gold, coal and uranium mines, amongst others. Drainage water from these mines has serious effects on receiving water quality (both surface and ground water) and thus on aquatic ecosystems. Water quality changes may include increases in total dissolved solids and total suspended solids (TDS, TSS), sulphate, hardness and trace metals, and reductions in dissolved oxygen and pH. The deleterious effects of this drainage water may be evident a significant distance from the source of pollution and for many years after the "source" has been removed (i.e. after the mine closed). River reaches immediately below mine effluents are often devoid of life, whilst reaches further away from the effluents normally have biotic communities characteristically low in diversity and richness, and dominated by a few tolerant species or groups. Recently rehabilitation of abandoned mines, removal of the sulphate component of the effluent and inhibition of bacterial oxidation of pyrite have received attention, so that the effects of acid mine drainage, both on receiving water quality and on aquatic ecosystem, can now be reduced.

19.1 INTRODUCTION

Mining operations result in increased quantities of heavy metals in streams that receive effluents from the mining operations. Clearly the biotas of these streams are subjected to elevated levels of both toxic and non-toxic metals, and so the species assemblages of all taxa from bacteria to fish reflect the responses of the individual taxa (e.g. Norris *et al.* 1981; Armitage & Blackburn1982; McCormick *et al.* 1994). Mining operations may also release a toxic mix of metals and ore-extracting substances such as cyanide. A less expected effect of mining operations, though, is the intense acidity of water draining from mining sites. This issue is discussed below.

19.2 EFFECTS OF MINING

The main operations in South Africa are gold, uranium and coal mining. Acid mine drainage from these operations may have a considerable impact on aquatic ecosystems.

19.2.1 Acid mine drainage

South Africa has extensive gold, uranium and coal mines, drainage from which potentially affects many of our water bodies. Mine effluents contain a complex of chemicals, many of which may have deleterious effects upon aquatic ecosystems. Chemical changes taking place in receiving waters include increased levels of suspended solids (increased siltation), dissolved solids, hardness, sulphates and trace metals; and reduction in dissolved oxygen concentration and pH (Clarke 1974). (The effects of each of these variables on aquatic ecosystems are detailed in separate chapters). The serious effect of mine effluents on water quality in rivers and lakes is well documented (Harrison 1962; Besch *et al.* 1972; Dills & Rogers 1974; Brown 1977; Ward *et al.* 1978; Moon & Lucostic 1979; Moore 1980; Scullion & Edwards 1980; Chadwick *et al.* 1986; Lampkin & Sommerfeld 1986; Filipek *et al.*

1987; Johnson & Thornton 1987; Younger 2001). The formation of acid drainage water is described by Letterman & Mitsch (1978): it occurs when sulphur in a coal seam or surrounding strata is uncovered and exposed to the oxidizing action of air, water and chemosynthetic bacteria that utilize the energy obtained from the conversion of inorganic sulphur to sulphate and sulphuric acid. The overall chemical reaction is:

$$2FeS_2 + 7O_2 + 2H_2O \rightarrow 2FeSO_4 + 2H_2SO_4$$

In receiving streams the ferrous iron will be oxidized to the ferric form by chemical, and sometimes by biological, means. If the mine discharge results in acid conditions in the stream, this oxidation will be slow. If, however, the acid is neutralized (the rate of neutralization depends on the surface geology) and pH rises to between 7 and 8, the rate of oxidation will increase and ferric hydroxide will precipitate. A layer of ferric hydroxide precipitate, so-called "yellowboy", on stream bottoms and structures such as bridge piers is a common sight in areas affected by acid mine drainage. The overall effect is thus dependent on the nature of the receiving water body. The deleterious effects of mine drainage water may affect streams as far as 18 km downstream from the mine and often remain long after mining has ceased (Savage & Rabe 1973 cited by Chadwick *et al.* 1986). Even underground the rocks exposed by mining operations may be affected in the same way, resulting in accumulation of extremely acid water. The Blesbok Spruit in the Gauteng area is a well known local example of a surface stream being contaminated by pumping acid minewater into a surface stream.

Ward *et al.* (1978) compared the impact of mining in the western and eastern United States, and concluded that the effect of mining was considerably less in the west. This difference was related to differences in the buffering capacity of streams in the two areas, western streams being highly buffered. Areas affected by mine effluents usually have communities dominated by chironomine and tendipedid midges, oligochaetes (worms), chaoborids (ghost midges) or tendipedids (midges), whilst odonates (dragonflies and damselflies), ephemeropterans (mayflies), bivalves and gastropods are absent (Clarke 1974, Letterman & Mitsch 1978, Moon & Lucostic 1979). Interestingly, trichopterans (caddisfliy larvae) of the genus *Hydropsyche* sp. have been found by both Brown (1977) and Letterman & Mitsch (1978) to be tolerant of mine drainage, possibly because of the silken, food-capturing net that they spin around themselves.

19.2.2 Gold and uranium mining

Förstner & Wittmann (1981) present summarized data (Table 19.1) on the composition of various effluents from gold and uranium mines in the Witwatersrand area of South Africa. Drainage occurs from the mine itself and from waste rock dumps and tailing areas, which contain large quantities of the sulphides and/or sulphates that are associated with most ore and coal bodies. The high sulphate and low pH levels result in the formation of hydrogen sulphide. The most commonly occurring sulphides are those of iron, namely the minerals pyrite, pyrrhorite and marcasite. It has been established that insidious seepage from gold/uranium slime dams is characterized by extremely low pH values, high sulphate contents and elevated trace metal concentrations (Table 19.1). Harrison (1962) reports on two rivers (the Klipspruit and Klip) in the Witwatersrand area. Characteristic values of the Klip River were: pH = 3.7-4.8, TDS = 930-1530 mg l^{-1} and SO_4 = 405-1660 mg l^{-1}. The fauna was impoverished, consisting of wide-spread acid-resistant species such as *Hydrozetes* sp. (an oribatid mite), the larvae of *Argyrobothrus* sp. (a hydroptilid caddisfly) and chironomid midges (Harrison 1962).

19.2.3 Coal mining

Pollution of surface waters by acidic drainage from working and disused coal mines has a significant impact on aquatic ecosystems. The various negative effects are succinctly described by Kemp (1967): "Coal mine drainage adversely affects the aesthetic appearance of streams and rivers, destroys the living organisms that inhabit them and hence reduces their self-purification power, and makes streams unfit for domestic, industrial and agricultural use, requiring surface waters to be extensively treated before they are suitable for such uses". Receiving water from acidic coal mine drainage is normally colourless to deep orange, contains sulphates of ferrous and ferric iron, aluminium, calcium, magnesium and usually sodium, has a very low pH (down to 2) and a high TDS. Table 19.2 clearly shows the differences in various water quality variables between polluted and unpolluted streams.

Table 19.1 Hydrochemistry of effluents from gold-mining wastes (mg l^{-1}) (from Förstner & Wittmann 1981) (N = normal river water, KD = Klerksdorp, WW = West Wits, WR = West Rand, CR = Central Rand, ER = East Rand, EV = Evander, OFS = Orange Free State).

	pH	SO_4	Fe	Mn	Zn	Cu	Pb	Cr	Ni	Cd	Co
N	6-9	0.011	0.1	0.007	0.01	0.003	0.0005	0.001	0.001	0.0005	0.0002
KD	2.7	2.70	110	14	9.00	1.6	0.020	0.70	6.80	0.023	3.90
WW	1.7	11.13	550	206	26.00	5.4	0.290	4.00	6.40	0.052	3.30
WR	3.7	2.95	3	18	4.68	1.22	0.081	0.02	1.54	0.007	2.25
CR	3.0	4.50	124	41	13.20	0.54	0.032	0.072	15.90	0.0061	2.61
ER	4.0	0.34	-	4	1.15	0.06	-	-	1.92	0.0002	0.27
EV	3.2	2.96	29	10	3.00	0.40	0.002	0.061	3.80	0.0025	1.50
OFS	2.6	6.53	274	44	5.22	2.32	0.010	0.060	8.90	0.005	2.06

Various studies (Edwards *et al.* 1972, Learner *et al.* 1971, Scullion & Edwards 1980) found streams downstream of coalfields dominated by chironomids (subfamily Orthocladiinae) and oligochaetes, mainly tubificids and naidids. Simuliids and trichopterans were vitually eliminated. Macroinvertebrate faunal abundance was decreased by 80-90% in a small stream in South Wales receiving coal industry pollutants (Scullion & Edwards 1980). Harrison (1962) measured conditions in the Klipspruit River, Witbank, which receives runoff from coalfields, and noted a pH of 2.9, TDS of 241-624 mg l^{-1}, and SO_4 of 475mg l^{-1}. The fauna was impoverished, and consisted mostly of *Hydrozetes* sp. and other oribatid mites, the caddis *Oxyethira velocipes* and the chironomid *Polypedilum anale*. No fish were found in these streams. Some of the more widespread species found in naturally acidic Cape streams have been able to colonize the acid-polluted Transvaal streams. Species richness is impoverished but densities are not necessarily low.

If precipitates of iron are deposited on the leaves of macrophytes and on the surfaces of algae, photosynthesis may be inhibited resulting in the disappearance of these communities in the river ecosystem (Hellawell 1986). Waste solids (< 64 μm in size and in test concentrations of 50, 100 and 200 mg l^{-1}) from a coal washery led to a reduction in the growth of rainbow trout, *Salmo gairdneri*, probably through reduced foraging and respiratory efficiency (Herbert & Richards 1963, cited by Hellawell 1986), although actual survival was not affected. The impact of mine drainage on a receiving stream is greater if the water body is small and the flow rate is low.

Oliff (1963) described conditions in Natal coalfields. Acid sulphates were produced but were neutralized rapidly by carbonates in the surface rocks and soils. Streams then carried large concentrations of the sulphates of calcium and magnesium. Oliff reported increased algal growth below the neutralization zone, probably as a result of the increased nutrient salts coming from the coalmines. The increased algal growth lead to increased faunal density and changes in species composition. The pH was between 2 and 4, TDS 2500-3000 mg l^{-1} and sulphates approximately 2000 mg l^{-1}. Swift (1985) further noted that coal piles stored near mine sites also have an effect on receiving water quality and aquatic communities.

Table 19.2. Analyses of polluted and unpolluted streams and effluents from Natal Coalfields (modified from Kemp 1967)(concentrations in mg l^{-1}).

Source/Stream	Type	pH	TDS	$CaCo_3$	Ca	Mg	Na	K	SO_4	Cl	SiO_2
Upper Umzinyatshana	Unpolluted stream	8.6	232	173	26.6	20.8	23.7	1.3	7.4	4.6	43.3
Upper Sundays	Unpolluted stream	8.2	95	79	11.1	7.2	8.2	1.5	3.2	0.1	16.4
Ballengeich Colliery dump	Active mine	5.6	2110	-	290	79	116	7.3	1590	11.5	22
Ballengeich Colliery	Pump effluent	3.5	4365	-	482	309	202	11.3	3080	6.9	40
Newcastle Colliery	Abandoned mine	3.3	3000	-	311	157	33	5.3	2380	4.0	40
Lower Umzinyatshana	Polluted stream	7.4	1037	128	94.6	45.1	96	2.3	697	20.0	4.5
Wasbank	Polluted stream	8.2	3105	107	249	135	621	7.5	2010	12.9	5.5

19.3 CURRENT RESEARCH

Heightened awareness of the effects of mine waters (both from operational and abandoned mines) on receiving water quality (ground and surface water) has prompted studies on the rehabilitation of mining waste impounds (Wells 1987; Levy *et al.* 1987; Jewaskiewitz & Lombard 1987). Rehabilitation must aim at producing a stable environment on the deposit which will be self sustaining, non-polluting and aesthetically acceptable. This is achieved by reducing air and water pollution to acceptable levels, mainly through the use of water and vegetation (Wells 1987). The significance of mineral pollution from mining activities is clearly evident from the report on mineral pollution in the Vaal Barrage (Jones *et al.* 1989). Jones and co-workers ascertained that seepage from mine deposits (both slimes dams and sand dams) was probably the source of high salt loads in the Vaal River. Diffuse sources (including mine residues) accounted for 50% of the overall pollution load of the Vaal Barrage.

Recent work funded by the Water Research Commission (WRC 1990) has investigated the chemical removal of sulphates (Water Research Commission Report No 203/1/90, 1990), and the inhibition of bacterial oxidation of pyrite and concomittant acid mine drainage (from coal waste dumps and gold sand dumps) (Loos *et al.* 1990a; Loos *et al.* 1990b; Sanderson & Immelman 1990).

It is clear therefore that the problem of water pollution by mine water drainage is receiving attention and this is to be encouraged to allow maintenance of appropriate conditions in aquatic ecosystems affected by mining activities, old and new. Brown *et al.* (2002, *in* Wood 2002) have recently produced a book detailing aspects of remediation of the environmental effects of mines and mine seepage, while authors such as Vinyard (1996) report on the development of new biological methods for assessing the effects of mines on aquatic ecosystems and the effectiveness of attempts at restoration.

CHAPTER 20: URBAN RUNOFF

SUMMARY

High population densities in urban areas and associated pollution problems place pressure on resources such as fresh water. Pollution from urban areas is usually from non-point sources. The type of contamination depends on runoff area (i.e. agricultural, residential, industrial) and pollutants may include a combination of physical (e.g. suspended solids), chemical (e.g. dissolved oxygen), organic (e.g. animal faecal material), inorganic (e.g. nutrients) and toxic (e.g. trace metals, biocides) constituents. Potential sources of urban runoff include road pavement materials, motor vehicles, atmospheric fallout, litter, vegetation, spills, domestic spraying, and unauthorized dumping and washing. Urbanization is characterized by an increased extent of impervious area (e.g. buildings and road surfaces), decreased natural storage capacity and canalisation of rivers into drainage canals, all of which lead to increased runoff volume and discharge rates. This alteration of flow regimes, together with chemical changes, results in the development in urban rivers of macroinvertebrate communities that are resilient enough to withstand intermittent periods of severe environmental stress such as the high flow rates, unstable substrata and high acute pollution loads.

20.1 INTRODUCTION

Migration of rural-dwellers to urban areas exerts increasing pressure on resources within these urban areas. One resource particularly susceptible to stress in urban areas is fresh water. Runoff in urban areas, particularly after heavy rainfall, is likely to contain pollutants that will ultimately make their way into local streams and rivers. Urban runoff may affect aquatic biotas (particularly faunal diversity) more severely than would well treated sewage effluent (Hellawell 1986) and is considered a major source of trace metals to aquatic environments in Australia (Liston & Maher 1986). To illustrate the potential impact of urban runoff, various studies have compared stormwater/urban runoff with treated sewage effluent (Weeks 1982). Stormwater runoff contained twenty times the concentration of non-filterable residues, twice the biological oxygen demand, fifteen times the phosphate and nitrogen concentrations, and more suspended solids than well-treated sewage effluents. Runoff from urban areas is diffuse, i.e. it is of non-point-source origin and is thus very difficult to quantify. It may include numerous pollutants (physical, chemical, organic, inorganic, toxic, non-toxic) that may interact to increase or decrease the ultimate effects on aquatic ecosystems. The actual constituents found in urban runoff will depend to a certain extent on runoff areas: for instance herbicides may be found in runoff in residential urban areas, trace metals in industrial urban areas and suspended solids in urban areas under construction. Within a catchment the following subsections based on land-use types may be distinguished (from Rimer *et al.* 1978):

a) Low activity rural: predominantly forested and agricultural without major roads.
b) High-activity rural: predominantly forested and agricultural but containing some major development, such as roadways and airports.

c) Low-activity residential: predominantly low-density residential with little or no commercial activity and few major thoroughfares.

d) High-activity residential: predominantly residential but with a heavier concentration of commercial and institutional areas and considerable vehicular traffic.

e) Low-activity commercial: predominantly light industry or commercial areas, where the major vehicular traffic occurs during the morning and afternoon rush-hour periods.

f) High-activity commercial: predominantly industrial, institutional or commercial, areas where vehicular traffic is likely to be heavy throughout the day.

g) Urban: predominantly central business districts.

Although a) and b) are more associated with non-urban areas they are included here, since within an urban area there are often open, non-developed areas like parks, sports fields and natural vegetation ("green belts"). In general, effects of urban runoff become more pronounced as areas develop from rural to residential to commercial (industrial/business district zone) and as traffic volume increases (Helsel *et al.* 1979; Stephenson & Green 1988).

Urban runoff is correlated to rainfall events and total runoff volumes are directly related to total rainfall (Weeks 1982). Urbanization of rural areas is characterized by an increase in impervious areas (e.g. buildings and road surfaces), decreased natural storage capacity, and canalisation of rivers into drainage canals. These factors result in increased runoff volume and discharge rates with catchment imperviousness and drainage infrastructure considered the primary determinants of the magnitude of stormwater runoff impacts (Walsh 2000). A storm of average intensity may produce 5-10 times the dry-weather flow contribution from the same area (Murphy & Carleo 1978). Generally low flows are lower and high flows higher than under natural conditions, because of the reduced contribution of ground water and the reduced interchange between surface and ground water (impervious layer). The "initial flush" at the start of a rainfall event transports approximately 40% of pollutants (Weeks 1982). In highly seasonal areas, such as the southwestern Cape, South Africa, the effects of the first flush must be highly significant.

20.2 POLLUTANTS CONTRIBUTING TO URBAN RUNOFF

An urban environment has many potential sources of contaminants reaching runoff (Table 18.1) from direct washing of the ground (e.g. road surfaces and roofs) or through fallout from industries and vehicle exhausts. Natural debris also forms part of the load of material in urban runoff. Over a long period (e.g. a season or a year) almost all of the pollution deposited on impervious surfaces that has not been removed by street cleaning, wind or decay, will eventually end up in surface runoff (Novotny & Chestes 1981).

20.3 EFFECTS OF URBAN RUNOFF ON AQUATIC ECOSYSTEMS

Some of the numerous adverse effects on receiving waters from urban runoff include physical effects such as flooding, erosion, sedimentation; physico-chemical effects such as elevated temperatures, dissolved oxygen depletion, nutrient enrichment, toxicity; and biological effects such as reduced biodiversity (Marsalek *et al.* 2002).

Table 20.1 Potential non-point sources of pollutants contributing to urban runoff

Potential sources	Examples
Road pavement materials	Degradation products
Motor vehicles	Fuel, lubricants, hydraulic fluids, coolants, linings, exhaust emissions, rust, vehicle components
Atmospheric fallout	From industrial stacks and vents
Litter	Packaging materials, food discards, animal droppings, plant debris
Vegetation	Bark, twigs, leaves, fruit, seeds, pollen, grasses
Spills	Sand, gravel, cement, agricultural and petroleum products, etc.
Domestic spraying	Herbicides, insecticides
Unauthorized dumping or washing	Chemicals such as detergents and oils

20.3.1 Physical changes

The impervious nature of urban areas translates into increased risk of urban flooding. Because of the increased flood risks, many urban channels have been modified to improve hydraulic efficiency, through straightening or sealing, or construction of levees resulting in isolation from floodplains (Walsh 2000). Riparian vegetation if often cleared for the same reasons. Canalisation (also referred to as channelisation) of a stream or river effectively destroys most natural processes occurring in it. In extreme cases of canalisation natural stream biotopes (riffle, run, pool, sand, backwater, marginal vegetation) are replaced by one "biotope": a smooth-sided and -bottomed concrete drain. Aquatic organisms adapted to living in the hyporheos (streambed sand/gravel) are effectively cut off from the substratum, as are processes such as filtration and seepage, which are important for the maintenance of healthy riparian vegetation. The physical changes employed to reduce hydrological effects such as flooding, reduce the diversity and availability of instream habitat for aquatic plants and animals (Walsh 2000). Increased discharge rates and flood events take on new meaning for aquatic organisms with no means of escape. The erosional/depositional characteristics, which often lead to increases in the concentration of suspended solids, are altered. The ability of a stream to self-purify depends on the biotic component, a reduction in which would reduce this ability. Clearing of riparian vegetation is often associated with canalisation and it may lead to changes in stream temperature regime and primary food source.

Alternative, slightly more "river-friendly" methods of canalisation include the use of Loffelstein blocks, which all the development of some biological communities in the interlocking blocks but which, by their size and design, tend to prevent the growth of riparian trees (Davies & Day 1998). Rehabilitation of canalised urban rivers via the introduction of small weirs and drilling of one-metre-diameter cores that facilitated the reconnectivity of the river to its hyporheos, allowed some of the natural functioning of the river and its biota to redevelop (Davies & Day 1998). Whilst the enhancement of habitat complexity in an urban stream in Cape Town, South Africa (Davies & Day 1998) led to an increase in species diversity, an experimental assessment of increasing habitat complexity showed little change in species composition (Walsh & Breen, in press). Walsh & Breen (in press) concluded that the success of physical habitat restoration in urban streams was likely to be dependent on the degree of catchment imperviousness and drainage intensity. Walsh (2000) sugested that for streams receiving urban runoff, the most appropriate primary restoration actions are likely to be the retention and treatment of stormwater before it reaches the streams.

20.3.2 Physico-chemical changes

The chemical impact of urban runoff on receiving water bodies depends on the types of pollutants that have been accumulated prior to being washed into the stream or lake. Urban runoff may result in a reduction in dissolved oxygen concentration (following organic enrichment: Rimer *et al.* 1978); an increase in nutrients (Bedient *et al.* 1980; Jones & Redfield 1984, Carpenter *et al.* 1998), trace metals (Wilber & Hunter 1979; Liston & Maher 1986; Hall & Anderson 1988, Marsalek *et al.* 2002), pesticides (Murphy & Carleo 1978) and hydrocarbons (Helsel *et al.* 1979; Hunter *et al.* 1979; Hoffman *et al.* 1982). Many urban pollutants are diffuse in origin, for example atmospheric deposition of nitrogen (Carpenter *et al.* 1998), and thus difficult to control. Details of the effects that changes in these factors (excluding hydrocarbons) have on aquatic ecosystems are reviewed in separate chapters.

The term "total hydrocarbons" refers to the sum of hydrocarbons in soluble and particulate fractions, the majority of which are petroleum hydrocarbons (Hoffman *et al.* 1982). In urban environments they are derived from accidental spills, deliberate dumping and vehicle crankcase oil. The highest concentration of petroleum hydrocarbons, which are largely associated with particulate matter, was linked to the first major peak in flow rate (Hoffman *et al.* 1982). An infrared method of oil quantification found that runoff from a busy highway contained an average concentration of 10 mg l^{-1} of oil, while runoff draining a residential area contained an average of 0.9 mg l^{-1} (Hoffman *et al.* 1982).

20.3.3 Biological changes

Recent studies have shown a relationship between the degree of catchment urbanisation, as assessed by catchment imperviousness, and ecological degradation (May *et al.* 1997, Walsh et al. in press). Biotic communities of streams showed increased degradation with increased catchment urbanisation (Walsh 2000). Walsh (2000) suggests that it is likely that severe degradation in the Melbourne metropolitan area (Australia) was a result of the efficiency of pollutant delivery to receiving streams in the metropolitan area. The traditional practices of stormwater management, aimed at reducing flood risk, maximised the efficiency of delivery to receiving waters, by maximising the connection between impervious surfaces and the receiving stream. Walsh (2000) further suggests that contemporary best management practices (BMPs) almost all aim to minimise connection, by retaining runoff for treatment and filtration.

Clearing of riparian vegetation reduces the amount of allochthonous material entering a stream, thereby affecting the primary food source and alters the flow regimes (e.g. increases in discharge rates and peaks) The causal relationship between physical and chemical changes, and changes in the resource base (particulate organic matter - POM) of an urban stream was investigated by Sloane-Richey *et al.* (1981). Food quality and temporal availability of POM were significantly lower in an urban stream (Kelsey Creek, Washington: 54% residential, 24% commercial and industrial, 22% park or undeveloped land) than in relatively undisturbed stream (Bera Creek, Washington: 15% residence, 85% forested or undeveloped pastureland). The altered flow regime influenced the resource base by increasing throughput of material and reducing residence time and instream-processing time for allochthonous and autochthonous material (Richey 1982 cited by Pedersen & Perkins 1986).

Accumulation of trace metals in urban runoff leads to a reduction in faunal density, diversity and richness, and a change in species composition (Garie & McIntosh 1986). Macroinvertebrate communities that develop in urban areas must be resilient enough to withstand intermittent periods of severe environmental stress such as high flow rates, unstable substratum and high acute pollution loads.

It is generally recognized that the overall effect of urbanization and stormwater runoff is debilitation of the functioning of the aquatic ecosystem. Macroinvertebrate and fish populations become depleted, algal biomass increases (resulting from nutrient enrichment) and bacterial decomposition pathways become important (Sloane-Richey *et al.* 1981). From a bioassessment perspective, Walsh (2000) suggests that, if the catchment of the impact study site is >25% impervious, macroinvertebrates are unlikely to be a useful group for impact assessment. He suggests that in such cases, monitoring the loads of the pollutants of interest may be a more appropriate approach. This conclusion warrants consideration by authorities currently involved in biomonitoring activities within urban catchment areas such as the Cape Metropolitan Council (CMC), South Africa.

Increased concerns about the impact of urban runoff, storm water in particular, has led to the implementation of "best management practices" (BMPs; Marsalek *et al.* 2002). Among the BMPs, stormwater management ponds are used to control flow and thereby reduce runoff peaks, and enhance stormwater quality by various physical, chemical and biological processes (Marsalek *et al.* 2002).

ASHMORE P., CONLY F.M., deBOER D., MARTIN Y., PETTICREW E. & ROY A. 2000. Recent (1995-1998) Canadian research on contemporary processes of river erosion and sedimentation, and river mechanics. *Hydrob. Process.* **14**: 1687-1706.

ASTON R.J. 1973. Field and experimental studies on the effects of a power station effluent on Tubificidae (Oligochaeta: Annelida). *Hydrobiol.* **42**: 225.

AUSTIN B. 1985. Antibiotic pollution from fish farms: effects on aquatic microflora. *Microbiol Sci.* **2**: 113-117.

*AZEVEDO J. & STOUT P.R. 1978. *Farm animal manures: an overview of their role in the agricultural environments.* Manual 44, California Agricultural Experimental Station Extension Service, CA, USA.

BACKES C.A. & TIPPING E. 1987. Aluminium complexation by an aquatic humic fraction under acidic conditions. *Wat. Res.* **21**: 211-216.

BAKER J.B. & SCHOFIELD C.L. 1982. Aluminium toxicity to fish in acidic waters. *Wat. Air & Soil Poll.* **18**: 289-309.

BARBOUR M.T., GERRITSEN J. & GRIFFITH G.E. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. *J. N. Am. Benthol. Soc.* **15** (2): 185-211.

BARBOUR M.T., GERRITSEN J., SNYDER B.D. & STRIBLING J.B. (1999). *Rapid Bioassessment Protocols for Use in streams and wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish*, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency, Office of Water, Washington, D.C., U.S.A.

BARLIN-BRINCK M. 1991. *Pesticides in southern Africa - an assessment of their use and environmental impact.* Wildlife Society of South Africa, Durban. 97pp.

BARRON M.G. 1990. Bioconcentration. *Environ. Sci. Technol.* **24**: 1612-1618.

BARTON B.A. 1977. Short-term effects of highway construction on the limnology of a small stream in southern Ontario. *Freshwater Biol.* **7**: 99-108.

*BARTON B.A., WHITE D.A. & PETERS E.J. 1972. *The effects of highway construction on the fish habitat in the Weber River, near Henefer, Utah.* Eng. Res. Cent., Colorado, Bur. Reclaim., Rep No. REC-ERC-72-17, June, 1972.

BAYLEY I.A.E. & WILLIAMS W.D. 1973. *Inland waters and their ecology.* Longmans, Australia.

BEADLE L.C. 1981. *The inland waters of tropical Africa: an introduction to tropical limnology.* Longmans, London.

BEDIENT P.B., LAMBERT J.L. & SPRINGER N.K. 1980. Stormwater pollutant load-runoff relationships. *J. Wat. Poll. Contr. Fed.* **52**: 745-751.

BEKKER H. & BROWN C.A. 1995. Developing environmental guidelines for freshwater aquacultural operations: a case study. *Water SA* **21** (4): 379-384.

BELL H.L. 1971. Effect of low pH on the survival and emergence of aquatic insects. *Wat. Res.* **5**: 313-319.

BELL H.L. & NEBEKER A.V. 1969. Preliminary studies on the tolerance of aquatic insects to low pH. *J. Arkansas Entom. Soc.* **42**: 230-236.

BENSON W.R. 1969. The chemistry of pesticides. *Ann. New York Acad. Sci.* **160**: 7-

BERENZEN N., SCHULZ R. & LIESS M. 2001. Effects of chronic ammonium and nitrate contamination on the macroinvertebrate community in running water microcosms. *Wat. Res.* **35**(14): 3478-3482.

BERGHEIM A. & SELMER-OLSEN A.R. 1978. River pollution from a large trout farm in Norway. *Aquaculture* **14**: 267-270.

BERGHEIM A., HUSTVEIT H., KITTELSEN A. & SELMER-OLSEN A.R. 1984. Estimated pollution loadings from Norwegian Fish Farms. II. Investigations 1980-1981. *Aquaculture* **36**: 157-168.

BERGHEIM A., SIVERTSEN A. & SELMER-OLSEN A.R. 1982. Estimated pollution loadings from Norwegian fish farms. 1. Investigations 1978-1979. *Aquaculture* **28**: 347-361.

BERNARD D.P., NEILL W.E. & ROWE L. 1990. Impact of mild experimental acidification on short term invertebrate drift in a sensitive British Columbian Stream. *Hydrobiol.* **203**: 63-72.

BERRILL M., TAYLOR G.& SAVARD H. 1991. Are chloride cells involved in low pH tolerance and sensitivity? The mayfly possibility. *Can. J. Fish. aquat. Sci.* **48**: 1220-1225.

BESCH W.K., RICARD M. & CANTIN R. 1972. Benthic diatoms as indicators of mining pollution in the northwest Miramichi River systems, New Brunswick, Canada. *Int. Revue Ges. Hydrobiol.* **57**: 39-74.

BIESINGER K.E. & CHRISTENSEN G.M. 1972. Effects of various metals on survival, growth, reproduction and metabolism of *Daphnia magna*. *J. Fish. Res. Bd. Canada* **29**: 1691-1700.

BINKLEY D., BURNHAM H. & LLEN H.L. 1999. Water quality impacts of forest fertilisation with nitrogen and phosphorus. *Forest Ecol. Manag.* 121: 191-213.

BIRGE W.J., BLACK J.A., WESTERMAN A.G. & HUDSON J.E. 1980. Aquatic toxicity tests on organic elements occurring in oil shale. In: *Oil shale symposium: sampling, analysis and quality assurance*. United States Environmental Protection Agency Report No. EPA 600/9-80-022.

BLOXHAM P.F. 1999. The management of agricultural manures. *J. Roy. Agric. Soc. Eng.* **160**: 138-155.

BLYTH J.D. 1986. The influence of particulate matter on stream invertebrates - case studies of Dartmouth and Thomson dams. In: *Water quality management: the role of particulate matter in the transport and fate of pollutants*. (Ed. Hart B.T.). Water Studies Centre, Chisholm Institute of Technology, Melbourne, Victoria, Australia.

BOHM M. 1985. Rain acidity in Stellenbosch. *S. Afr. J. Sci.* **81**: 652-653.

BOND G.W. 1946. A geochemical survey of the underground water supplies of the Union of South Africa with particular reference to their utilisation in power production and industry. *Dept. of Mines Geological Survey Mem.* **41**. 216pp. Government Printer, Pretoria.

BONFORTE G., GUZZI L., MARTINOTTO W. & QUEIRAZZA G. 1986. Identification and choice of a biological indicator for the study of the contamination of a river by heavy metals. *Chem. in Ecol.* **2**: 199-204.

BORGMANN U. *et al.* 1989. Toxicity test procedures for *Hyalella azteca* and chronic toxicity of cadmium and pentachlorophenol to *H.azteca*, *Gammarus fasciata* and *Daphnia magna*. *Arch. Environ. Contam. Toxicol.* **18**: 756.

BOSTROM B., PERSSON G. & BROBERG B. 1988. Bioavailability of different phosphorus forms in freshwater systems. *Hydrobiol.* **170**: 133-155.

BOULTON A.J. & BROCK M.A. 1999. *Australian freshwater ecology: processes and management.* Cooperative research Centre for freshwater ecology, Gleneagles Publishing, Australia.

BOURASSA N. & CATTANEO A. 1998. Control of periphyton biomass in Laurentian stream (Quebec). *J. N. Am. Benthol. Soc.* **17**: 420-429.

BOWEN H.J.M. 1966. *Trace elements in biochemistry*. Academic Press, New York.

BOYD C.E. 1982. Water quality management for pond fish culture. In: *Development in aquaculture and fisheries sciences*. **Vol 9**. Elsevier Science Publishers, The Netherlands.

BOYLE T.P., SMILLIE G., ANDERSON J.C. & BEESON D.R. 1990. A sensitivity analysis of nine diversity and seven similarity indices. *J. Wat. Poll. Contr. Fed.* **62**: 749-762.

BRENNER F.J. & MONDOK J.J. 1995. Nonpoint source pollution potential in an agricultural watershed in northwestern Pennsylvania. *Wat. Res. Bull.* **31**(6): 1101-1112.

*BREZONIK P.L. 1972. Nitrogen: sources and transformations in natural waters. Pp. 1-50. In: *Nutrients in natural waters*. (Eds Allen H.E & Kramer J.R.). John Wiley & Sons, New York.

BRINGMANN G. & KUHN R. 1959. The toxic effects of waste water on aquatic bacteria, algae and small crustaceans. *Gesundheits Ing.* **80**: 115.

BRINK D. & BEKKER H. 1991. *'n Verslag oor die potensiaal van akwakultuur in die Winterreenstreek*. Departement Landbou-ontwikkeling. Winterreenstreek, Elsenburg, South Africa. 22pp.

BRITTON D.L. 1991. Fire and the chemistry of a South African mountain stream. *Hydrobiol.* **218**: 177-192.

BRITTON D.L., DAY J.A. & HENSHALL-HOWARD M-P. 1992. Hydrochemical response during storm events in a South African mountain catchment: the influence of antecedent conditions. *Hydrobiol.* **250**: 143-157.

BRITTON D.L., DAY J.A. & HENSHALL-HOWARD M.-P. 1993. Hydrochemical responses during storm events in a South African mountain catchment: the influence of antecedent conditions. *Hydrobiol.* 250: 143-157.

BROCK M.A. 1985. Are Australian salt lake ecosystems different? Evidence from the submerged aquatic plant communities. *Proc. Aust. Ecol. Soc.* **14**: 43-50.

BROKOVIK-POPOVIC T. & POPOVIC M. 1977. Effects of heavy metals on survival and respiration rate of tubificid worms. Part 1: Effects of survival. *Environ. Poll.* **13**: 65-72.

BROWN B.E. 1977. Effects of mine drainage on the river Hayle, Cornwall. a) factors affecting the concentration of copper, zinc and iron in water, sediments and dominant invertebrate fauna. *Hydrobiol.* **52**: 221-233.

BROWN C.A. 1991. *A preliminary review of special effluent standards* (Amendments to the Water Act 1984). Freshwater Research Unit, University of Cape Town. Unpublished report to Department of Water Affairs and Cape Nature Conservation. 45pp.

BROWN C.A. 1992. *Initial surveys in the investigation to determine the effects of trout-farm effluent on riverine biotas in the south-western Cape.* Unpublished report to the Department of Water Affairs and Forestry, and the Cape Nature Conservation. 68pp.

BROWN C.A. 1996. Macroinvertebrate community patterns in relation to physico-chemical parameters measured at two land-based trout farms affecting streams in the south-western Cape, South Africa. *Arch. Hydrobiol.* **138** (1): 57-76.

BROWN C.A. 1997. *Modeling and managing the effects of trout farms on Cape rivers.* Unpublished Doctoral thesis, Zoology Department, University of Cape Town, South Africa. 175pp.

BROWN C.A. & KING J.M. 1995. The effects of trout-farm effluents on benthic invertebrate community structure in rivers in the south-western Cape, South Africa. *Sth. Afr. J. aquat. Sci.* **21** (1/2): 3-21.

BROWN C.A. & KING J.M. 2001. World Bank Water Resources and Environmental Best Management Briefs: Briefing Note 6. *Environmental Flow Assessments: Concepts and Methodologies.* 22p.

BROWN D.J.A. 1983. Effect of calcium and aluminium concentrations on the survival of brown trout (*Salmo trutta*) at low pH. *Bull. environ. Contam. Toxicol.* **30**: 582-587.

BROWN G.W. & KRYGIER T. 1970. Effect of clear-cutting on stream temperature. *Wat. Res.* **6**: 1133-1140.

BROWN V.M. 1982. *The use of toxicity data in environmental toxicology.* Environment and quality of life, International Symposium, Sopha Antipolis. CEC Luxembourg.

BROWN M., BARLEY B. & WOODS H. 2002. *Minewater treatment: technology, application and policy.* IWA Publishing.

BRUGGEMAN W.A. 1982. Hydrophobic interactions in the aquatic environment. In: *The handbook of environmental chemistry. Reactions and processes.* (Ed. Hutzinger O.). **Vol. 2**, Part B. Springer-Verlag, New York.

BRUNGS W.A. 1969. Chronic toxicity of zinc to the fathead minnows (*Pimephales promelas* Rafinesque). *Trans. Amer. Fish. Soc.* **98**: 272-279.

BRUNGS W.A. 1971a. Chronic effects of elevated temperature on the fathead Minnow (*Pimephales promelas* Rafinesque). *Trans. Am. Fish. Soc.* **100**: 659-664.

BRUNGS W.A. 1971b. Chronic effects of low dissolved oxygen concentrations on the fathead minnow (*Pimephales promelas*). *J. Fish. Res. Bd. Canada* **28**: 1119-1123.

BRUTON M.N. 1985. The effects of suspensoids on fish. *Hydrobiol.* **125**: 221-241.

*BRYAN G.W. 1976a. Heavy metal contamination in the sea. Pp. 185-302. In: *Marine Pollution*. (Ed. R. Johnston). Academic Press, London.

BRYAN G.W. 1976b. Some aspects of heavy metal tolerance in aquatic organisms. Pp. 7-34. In: *Effects of pollutants on aquatic organisms*. (Ed. Lockwood A.P.M.). Cambridge University Press, Cambridge.

*BUERMANN Y., DU PREEZ H.H., STEYN G.J., HARMSE J.T. & DEACON A. 1995. Suspended silt concentrations in the lower Olifants River (Mpumalanga) and the impact of silt releases form the Phalaborwa Barrage on water quality and fish survival. *Koedoe* **38**: 11-34.

BURROWS W.D. 1977. Aquatic aluminium: chemistry, toxicology and environmental prevalence. *CRC Crit. Rev. environ. Contr.* **7**: 167-216.

BUTLER G.C. 1978. *Principles of Ecotoxicology*. Scientific committee on problems of the environment, SCOPE 12. John Wiley & Sons, Chichester.

BYREN B.A. & DAVIES B.R. 1989. The effect of stream regulation on the physiochemical properties of the Palmiet River, South Africa. *Reg. Rivers Res. Manag.* **3**: 107-121.

CADBURY D. 1997. *The feminisation of nature*. Penguin, London. 303 pp.

CAIRNS J.Jr. & SCHEIER A. 1959. The effects of temperature and hardness of water upon the toxicity of zinc to the common bluegill (*Lepomis macrochirus*). *Notul. Nat.* **229**: 1-12.

CAIRNS J.Jr. & WALLER W.T. 1971. The use of fish movement patterns to monitor zinc. *Wat. Poll. Contr. Res. Ser.*, United States Environmental Protection Agency 18050 EDP 12/71: 55pp.

*CAIRNS J.Jr., LANZA G.R. & PARKER B.C. 1972. Pollution related structural and functional changes in aquatic communities with emphasis on freshwater algae and Protozoa. *Proc. Acad. nat. Sci. Philadelphia* **124**: 79-127.

CALLELY A.G., FORSTER C.G. & STAFFORD D.A. (Eds.). 1977. *Treatment of industrial effluents*. Hodder & Stoughton, London.

CAMARGO J.A. 1992. Temporal and spatial variations in dominance, diversity and biotic indices along a limestone stream receiving a trout farm effluent. *Water, air & soil Poll.* **63**: 343-359.

CAMPBELL I.C. 1978. A biological investigation of an organically polluted urban stream in Victoria. *Aust. J. Mar. Freshwat. Res.* **29**: 275-291.

CAMPBELL I.C. & DOEG T.J. 1989. Impact of timber harvesting and production on streams: a review. *Aust. J. Mar. Freshwat. Res.* **40**: 519-539.

CAMPBELL P.G.C. & TESSIER A. 1987. Metal speciation in natural waters: Influence of environmental acidification. *Adv. Chem. Ser.: Sources & Fates of aquatic pollutants* **216**: 185-207.

CANADIAN GUIDELINES. 1987. *Canadian water quality guidelines*. Prepared by the task force on water quality guidelines of the Canadian Council of Resource and Environmental Ministers, Canada.

CANTON, S.P. & W.D. VANDERVEER. 1997. Selenium toxicity to aquatic life: an argument for sediment-based water quality criteria. *Environ Toxicol. Chem.*. **16:** 1255-1259.

CAO Y., BARK A.W. & WILLIAMS W.P. 1997. Analysing benthic macroinvertebrate community changes along a pollution gradient: a framework for the development of biotic indices. *Water Research* **31** (4): 884-892.

CARDWELL R.D., FOREMAN D.G., PAYNE T.R. & WILBUR D.J. 1976. Acute toxicity of selenium dioxide to freshwater fishes. *Arch. environ. Contam. Toxicol.* **4**: 129-144.

CARLINE R.F., DEWALLE D.R., SHARPE W.E., DEMPSEY B.A., GAGEN C.J. & SWISTOCK B. 1992. Water chemistry and fish community responses to episodic stream acidification in Pennsylvania, USA. *Environ. Poll.* **78**: 45-48.

CARLISLE D.M. & CLEMENTS W.H. 1999. Sensitivity and variability of metrics used in biological assessments of running waters. *Environmental Toxicology & Chemistry* **18** (2): 285-291.

DE NICOLA GIUDICI M., MIGLIORE L., GAMBARDELLA C. & MAROTTA A. 1988. Effect of chronic exposure to cadmium and copper on *Asellus aquaticus* (L.) (Crustacea, Isopoda). *Hydrobiol.* **157**: 265-269.

DE VLAMING V., CONNOR V., DIGIORGIO C., BAILEY H.C., DEANOVIC L.A. & HINTON D.E. 2000. Application of whole effluent toxicity test procedures to ambient water quality assessment. *Environ Toxicol. Chem.* **19**(1): 42-62.

DEAN J.G., BOSQUI F.L. & LANOUETTE V.H. 1972. Removing heavy metals from waste water. *Environ. Sci. Technol.* **6**: 518-522.

DEFC (Department of the Environment and Forestry Commission). 1991. *Forests and surface water acidification.* Forestry Commission, Edinburgh, United Kingdom.

DEL ROSARIO R.B., BETTS E.A. & RESH V.H. 2002. Cow manure in headwater streams: tracing aquatic insect response to organic enrichment. *J. N. Am. Benthol. Soc.* **21**(2): 278-289.

DEMARCH B.G.E. 1988. Acute toxicity of binary mixtures of five cations (Cu^{2+}, Cd^{2+}, Zn^{2+}, Mg^{2+} and K^{+}) to the freshwater amphipod *Gammarus lacustris* (Sars.). *Can. J. Fish. aquat. Sci.* **45**: 625.

DEMAYO A., TAYLOR M.C., TAYLOR K.W & HODSON P.V. 1982. Toxic effects of lead and lead compounds on human health, aquatic life, wildlife, plants and livestock. *CRC Crit. Rev. environ. Contr.* **12**: 257-305.

DE MOOR F. 1981. A community of *Simulium* species in the Vaal River near Warrenton. Unpublished Doctoral Thesis, Rhodes University, Grahamstown, South Africa.

DENISEGER J., AUSTIN A. & LUCEY W.P. 1986. Periphyton communities in a pristine mountain stream above and below heavy metal mining operations. *Freshwater Biol.* **16**: 209-218.

DWA (DEPARTMENT OF WATER AFFAIRS). 1986. *Management of water resources of the Republic of South Africa.* Government Printers, Pretoria, South Africa.

DWAF (DEPARTMENT OF WATER AFFAIRS AND FORESTRY). 1991. *Water quality management proposals and strategies in the R.S.A.* Department of Water Affairs and Forestry. Government Printers, Pretoria, South Africa.

DWAF (DEPARTMENT OF WATER AFFAIRS AND FORESTRY). 1996. *South African Water Quality Guidelines. Volume 7: Aquatic Ecosystems.* Produced by the CSIR Environmental Services, Pretoria for the Department of Water Affairs & Forestry, Pretoria, South Africa.

DWAF (DEPARTMENT OF WATER AFFAIRS AND FORESTRY). 2000. *A protocol for acute toxicity testing using selected riverine invertebrates in artificial stream systems.* Version 1.0. Produced by Scherman P.A. & Palmer C.G., Centre for Aquatic Toxicology, Institute for Water Research, Rhodes University, Grahamstown, South Africa.

DIAMOND M., HIRST D., WINDER L., CRAWSHAW D.H. & PRIGG R.F. 1992. The effect of liming agricultural land on the chemistry and biology of the River Esk, north-west England. *Environ. Poll.* **78**: 179-185.

DICKENS C.W.S. & GRAHAM P.M. 1998. Biomonitoring for effective management of wastewater discharges and the health of the river environment. *Aq. Ecosys. Health Manag.* **1**: 199-217.

DICKENS C.W.S. & GRAHAM P.M. 2002. The South African Scoring System (SASS) Version 5 Rapid Bioassessment method for rivers. *African J. Aquat. Sci.* **27**(1): 1-10.

*DIESCH S.L. 1970. Disease transmission of waterborne organisms of animal origins. In: *Agricultural practices and water quality.* Iowa State University Press, Ames, IA, USA, 265-285.

DILLS G. & ROGERS J. 1974. Macroinvertebrate community structure as an indicator of acid mine pollution. *Environ. Poll.* **6**: 239-262.

DISTEFANO R.J., NEVES R.J., HELFRICH L.A. & LEWIS M.C. 1991. Response of the crayfish *Cambarus bartonii bartonii* to acid exposure in southern Appalachian streams. *Can. J. Zool.* **69**: 1585-1591.

DITORO D.M, ALLEN H.E., BERGMAN H.L., MEYER J.S., PAQUIN P.R. & SANTORE R.C. 2001. Biotic ligand model of the acute toxicity of metals. I. Technical basis. *Environ Toxicol.*

Chem. **20**: 2383-2396.

DODDS W.K. & WELCH E.B. 2000. Establishing nutrient criteria in streams. *J. N. Am. Benthol. Soc.* **19**(10): 186-196.

DODDS W.K., SMITH V.H. & ZANDER N. 1997. Developing nutrient targets to control benthic chlorophyll levels in streams: a case study of the Clark Fork River. *Wat. Res.* **31**(7): 1738-1750.

DOEG T.J. & KOEHN J.D. 1994. Effects of draining and silting a small weir on downstream fish and macroinvertebrates. *Regul. Rivers. Res. Manag.* **9**: 263-277.

DOUDEROFF P. & KATZ M. 1953. Critical review of the literature on the toxicity of industrial wastes and their components to fish - II. The metals as salts. *Sewage Ind. Wastes* **25**: 802-839.

DRATNAL E. & KASPRZAK K. 1980. The response of the invertebrate fauna to organic pollution in a well oxygenated karst stream exemplified by the Pradnik stream (Poland). *Acta Hydrobiol.* **22** 263-278.

DREWETT N. & ABEL P.D. 1983. Pathology of lindane poisoning and hypoxia in brown trout, *Salmo trutta* L. *J. Fish. Biol.* **23**: 373.

DRISCOLL C.T. & SCHECHER E.D. 1990. The chemistry of aluminium in the environment. *Environ. Geochem. Health* **12**: 28-49.

DRISCOLL C.T., BAKER J.P., BISOGNI J.J. & SCHOFIELD C.L. 1980. Effects of aluminium speciation in dilute acidified waters. *Nature* **284**: 161-164.

*DRISCOLL C.T., BAKER J.P., BISOGNI J.J. & SCHOFIELD C.L. 1984. Aluminum speciation and equilibria in dilute acidic surface waters of the Adirondack region of New York State. In: *Geological aspects of acid rain.* (Ed. Bricker O.B.). American Chemical Society, Ann Arbour, Michigan, U.S.A.

DU PLESSIS H.M. & VAN VEELEN M. 1991. Water quality: salinization and eutrophication time series and trends in South Africa. *S. Afr. J. Sci.* **87**: 11-16.

DU PREEZ H.H. & STEYN G.J. 1992. A preliminary investigation of the concentration of selected metals in the tissues and organs of the tigerfish (*Hydrocynus vittatus*) from the Olifants River, Kruger National Park, South Africa. *Water S.A.* **18**: 131-136.

DU PREEZ H.H. & VAN VUREN J.H.J. 1992. Bioconcentration of artrazine in the banded Tilapia, *Tilapia sparrmanii*. *Comp. Biochem. Physiol.* **101**: 631-655.

DU PREEZ H.H., BUERMANN Y. STEYN G.J. & SMIT L. 1996. The influence of suspended silt on oxygen consumption by the redbreast tilapia, *Tilapia rendalli* (Boulenger, 1896). *Hydrobiol.* **325**: 31-38.

DUFFUS J.H. 1980. *Environmental Toxicology*. Edwards Arnold Publishers, London. 164pp.

DURRETT C.W. & PEARSON W.D. 1975. Drift of macroinvertebrates in a channel carrying heated water from a power plant. *Hydrobiol.* **46**: 33-43.

DU TOIT J. 1995. Doing away with DDT. *On Track* May/June: 12-14.

DYE & JONES. 1978. *A review of eutrophication, nutrient regulation and the impacts of nutrients on water quality.* Florida Department of Environmental regulation. Technical Series **3**: 1-34.

EDWARDS C.A. 1973. *Environmental pollution by pesticides.* Plenum Press, London.

EDWARDS R.W., BENSON-EVARIS K., LEARNER M.A., WILLIAMS P. & WILLIAMS R. 1972. A biological survey of the River Taff. *J. wat. Poll. Contr. Fed.* **71**: 144-166.

EIFAC (EUROPEAN INLAND FISHERIES ADVISORY COMMISSION). 1973. Water quality for European freshwater fish: report on dissolved oxygen and inland fisheries, *EIFAC Technical Paper* **19**: Food and Agricultural Organisation, Rome.

EISLER R. 1971. Cadmium poisoning in *Fundulus heteroclitus* (Pisces: Cyprinodontidae) and other marine organisms. *J. Fish. Res. Bd. Canada* **28**: 1225-1234.

EISLER R. 1985a Mirex hazards to fish, wildlife, and invertebrates: a synoptic review. Contaminant Hazard Reviews, Report no. 1: *U.S. Fish. Wildl. Serv. Biol. Rep.* **85** (1.1): 1-37.

EISLER R. 1985b Carbofuran hazards to fish, wildlife, and invertebrates: a synoptic review. Contaminant Hazard Reviews, Report no. 3: *U.S. Fish. Wildl. Serv. Biol. Rep.* **85** (1.3): 1-36.

EISLER R. 1985c. Cadmium hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish. Wildl. Serv. Biol. Rep.* **85** (1.2). 46pp.

EISLER R. 1985d. Selenium hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish. Wildl. Serv. Biol. Rep.* **85** (1.5). 57pp.

EISLER R. 1986a. Diazinon hazards to fish, wildlife, and invertebrates: a synoptic review. Contaminant Hazard Reviews, Report no. 9: *US Fish and Wildlife Service Biological Report* **85** (1.9): 1-37.

EISLER R. 1986b. Chromium hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish. Wildl. Serv. Biol. Rep.* **85** (1.6). 60pp.

EISLER R. 1988a. Arsenic hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish. Wildl. Serv. Biol. Rep.* **85** (1.12). 92pp.

EISLER R. 1988b. Lead hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish. Wildl. Serv. Biol. Rep.* **85** (1.14). 134pp.

EISLER R. 1989a. Pentachlorophenol hazards to fish, wildlife, and invertebrates: a synoptic review. Contaminant Hazard Reviews, Report no. 17: *US Fish and Wildlife Service Biological Report* **85** (1.17): 1-37.

EISLER R. 1989b. Molybdenum hazards to fish, wildlife, and invertebrates: a synoptic review. *U.S. Fish. Wildl. Serv. Biol. Rep.* **85** (1.19). 61pp.

EISLER R. & JACKNOW J. 1985. Toxaphene hazards to fish, wildlife, and invertebrates: a synoptic review. Contaminant Hazard Reviews, Report no. 4: *US Fish and Wildlife Service Biological Report* **85** (1.4): 1-34.

ELDRIDGE E.F. 1960. *Return irrigation water, characteristics and effects.* United States Public Health Service, Region IX, Portland, Oregon.

EPA. 1980. United States Environmental Protection Agency. *Ambient water quality criteria for arsenic.* USEPA Rep. 440/5-80-021. 205pp.

EPA. 1985. United States Environmental Protection Agency. *Ambient water quality criteria for arsenic* - 1984. USEPA Rep. 440/5-84-033. 66pp.

EVANS G.H. & MARCAN E. 1976. Diatom community response to variations in effluent concentration. *Environ. Poll.* **10**: 115-126.

FAIRWEATHER P.G. 1999. State of environment indicators of "river health": exploring the metaphor. *Freshwat. Biol.* **41**: 211-220.

FILIPEK L.H., NORDSTROM D.K. & FICKLIN W.H. 1987. Interaction of acid mine drainage with water and sediments of West Squaw Creek in the west Shasta mining district, California. *Environ. Sci. Tech.* **21**: 388-396.

FINLAYSON B.J. & VERRUE K.M. 1982. Toxicities of copper, zinc and cadmium mixtures to juvenile chinook salmon. *Trans. Am. Fish. Soc.* **111**: 645-650.

FJELLHEIM A. & RADDUM G. 1992. Recovery of acid-sensitive species of Ephemeroptera, Plecoptera and Trichoptera in River Audna after liming. *Environ. Poll.* **78**: 173-178.

FLÜGEL W.-A. 1993. River salination due to non-point contribution to irrigation return flow in the Breede River, Western Cape Province, South Africa. *Wat. Sci. Technol.* **28**: 193-197.

FORE L. S., KARR J. R. & WISSEMAN R. W. 1996. Assessing invertebrate responses to human activities: evaluating alternative approaches. *J. N. Am. Benthol. Soc.* **15** (2): 212-231.

FÖRSTNER U. & WITTMANN G.T.W. 1981. *Metal Pollution in the aquatic environment.* Springer-Verlag, Berlin. 486pp.

FÖRSTNER U. & PROSI F. 1979. Heavy metal pollution in freshwater ecosystems. Pp. 129-161 *in*: O. Ravera (ed.) *Biological aspects of freshwater pollution.* Pergamon Press, Oxford.

FOSTER P.L. 1982. Metal resistance of Chlorophyta from rivers polluted by heavy metals. *Freshwater Biol.* **12**: 41-61.

FOURIE J.M. & GÖRGENS A.H.M. 1977. *Mineralisation studies of the Berg River (1974 to 1976).* Council for Scientific and Industrial Research. Report No. 334. Pretoria, South Africa. 30pp.

FOX M.J. & WILCOCK R.L. 1988. Soil, water, and vegetation residues of 2,4,5-T applied to a New Zealand hillside. *N. Z. J. agric. Res.* **31**: 347-357.

FRASER J. 1980. Acclimation to lead in the freshwater isopod *Asellus aquaticus*. *Oecologia* **45**: 419-420.

FURSE M.T. 2000. The application of RIVPACS procedures in headwater streams - an extensive and important natural resource. In: *Assessing the biological quality of fresh waters: RIVPACS and other techniques* (Eds. Wright J.F., Sutcliffe D.W. & Furse M.T.), Freshwater Biological Association, United Kingdom.

GALE B.A. 1992. *The effect of regulation by two impoundments on an acid, blackwater, Cape mountain stream.* Unpublished Ph.D Thesis, University of Cape Town, South Africa. 272pp.

GALPIN J.S. & TURNER C.R. 1999. Trends in rain quality data from the South African interior. *S. Afr. J. Sci.* **95**: 223-225.

GAMMETER S. & FRUTIGER A. 1990. Short-term toxicity of ammonia and low oxygen to benthic macroinvertebrates of running waters and conclusions for wet weather water pollution control measures. *Wat. Sci. & Tech.* **22**: 291-296.

GARDINER A.J.C. 1988. *A study of the water chemistry and plankton in blackwater lakelets in the south-western Cape, South Africa.* Unpublished Doctoral Thesis, University of Cape Town, South Africa. 240pp.

GARDINER J. & ZABEL T. 1989. *United Kingdom water quality standards arising from European Community directives-An update.* Water Research Centre Report. No. PRS 2287-M. 120pp.

GARIE H.L. & MCINTOSH A. 1986. Distribution of benthic macroinvertebrates in a stream exposed to urban runoff. *Wat. Res. Bull.* **22**: 447-455.

GENIS N de L. & RABIE M.A. 1983. Biocides. Pp. 412-434. In: *Environmental concerns in South Africa: technical and legal perspectives.* (Eds. Fuggle R.F. & Rabie M.A.). Juta & Co. Ltd., Cape Town.

GENTER R.B., CHERRY D.S., SMITH E.P. & CAIRNS J.Jr. 1987. Algal-periphyton population and community changes from zinc stress in stream mesocosms. *Hydrobiol.* **153**: 261-275.

GENTER R.B. & AMYOT D.J. 1994. Freshwater benthic algal population and community changes due to acidity and aluminum-acid mixtures in artificial streams. *Environ Toxicol. Chem.*. **13**: 369-380.

GERHARDT A. 1993. Review of impact of heavy metals on stream invertebrates with special emphasis on acid conditions. *Wat. Air Soil Poll.*. **66**: 289-314.

GERRITSEN J., BARBOUR M.T. & KING K. 2000. Apples, oranges, and ecoregions: on determining pattern in aquatic assemblages. *J. N. Am. Benthol. Soc.* **19** (3): 487-496.

GERSICH F.M. & HOPKINS D.L. 1986. Site-specific acute and chronic toxicity of ammonia to *Daphnia magna* Strauss. *Environ. Toxicol. Chem.* **5**: 443-447.

GIBBS R.J. 1970. Mechanisms controlling world water chemistry. *Science* **170**: 1088-1090.

GIESY J.P., LEVERSEE G.J. & WILLIAMS D.R. 1977. Effects of naturally occurring aquatic organic fractions of cadmium toxicity to *Simocephalus serrulatus* (Daphnidae) and *Gambusia affinis* (Poeciliidae). *Wat. Res.* **11**: 1013-1020.

GILIOMEE J.H. & GLAVOVIC P.D. 1992. Pesticides. Pp. 523-543. In: *Environmental management in South Africa* (Eds. Fuggle R.E. & Rabie M.A.). Juta & Co. Ltd., Wetton.

GIPPEL C.J. 1989. The use of turbiditimeters in suspended sediment research. *Hydrobiol.* **176/177**: 465-480.

GJESSING E.T., HENRIKSEN A., JOHANNESSEN M. & WRIGHT R.F. 1976. Effects of acid precipitation on freshwater chemistry. Pp 64-85. In: *Impact of acid precipitation on forest and freshwater ecosystems in Norway.* (Ed. Braekke F.W). SNSF Project Research Report 6/76, Oslo.

GLAZEWSKI J. 2002. *Environmental Law in South Africa.* Butterworths, Durban.

GOLDSBOROUGH L.G. & ROBINSON G.G.C. 1986. Changes in periphytic algal community structure as a consequence of short herbicide exposures. *Hydrobiol.* **139**: 177-192.

GOLTERMAN H.L. 1975. *Physiological Limnology.* Elsevier, New York. 489pp.

GOLTERMAN H.L. & KOUWE F.A. 1980. Chemical budgets and nutrient pathways. Pp. 85-140. In: *The functioning of freshwater ecosystems.* (Eds. Le Cren E.D. & McConnell R.H.). Cambridge University Press, Cambridge.

GOLTERMAN H.L., CLYMO R.S. & OHNSTAD M.A.M. 1978. *Methods for the physical and chemical analysis of fresh waters.* IBP Handbook No. 8. Blackwell Scientific Publications, Oxford.

GORHAM E. 1955. On the acidity and salinity of rain. *Geochim. Cosmochim. Acta* **7**: 231-239.

GORSUCH J. & KLAINE S. (eds) 1999. Annual Review Issue: Silver Toxicity. *Environ Toxicol. Chem.* **18:** 1-108.

GRABOW W.O.K., MORGAN W.S.G. & SLABBERT J.L. 1985. Bioassays used for evaluating the quality of reclaimed water in southern Africa. *Water Quality Bull.* **10**: 29-35,59.

GRAY L.J. & WARD J.V. 1982a. Effects of sediment releases from a reservoir on stream macroinvertebrates. *Hydrobiol.* **96**: 177-184.

GRAY L.J. & WARD J.V. 1982b. *Effects of releases of sediment from reservoirs on stream biota.* Completion Report No. 116. Colorado State University.

GRAYNORTH E. 1979. Effects of logging on stream environments and faunas in Nelson. *N. Z. J. mar. freshwater Res.* **13**: 79-109.

GRIB I.V. & KOMAROVSKIY F.Y. 1990. Expert evaluation of toxic pollution of freshwater ecosystems. *Hydrobiol. J.* **26**: 71-76.

GRIFFITH M.B., PERRY S.A. & PERRY W.B. 1995. Macroinvertebrate communities in headwater streams affected by acid precipitation in the central Appalachians. *J. Environ. Qual.* **24**: 233-238.

GRIMSHAW D.L. & FUGE J.L. 1976. Seasonal and short-term variations in the zinc concentration and supply of dissolved zinc to polluted aquatic environments. *Environ. Poll.* **11**: 1-7.

GROBLER D.F. 1994. A note on PCBs and chlorinated hydrocarbon pesticide residues in water, fish and sediment from the Olifants River, Eastern Transvaal, South Africa. *Water SA* **20:** 187-194.

GROBLER E., DU PREEZ H.H. & VAN VUREN J.H.J. 1989. Toxic effects of zinc and iron on the routine oxygen consumption of *Tilapia sparrmanii* (Cichlidae). *Comp. Biochem. Physiol.* **94**C: 207.

GROOM A.P & HILDREW A.G. 1989. Food quality for detritivores in streams of contrasting pH. *J. Animal Ecol.* **58**: 863-881.

GUCKERT J.B. 1996. Toxicity assessment by community analysis. *J. Microbiol. Methods* **25**: 101-112.

GUEROLD F. & PIHAN J.C. 1989. L'acidification des torrents vosgiens. Mise en évidence. Impact sur les populations de macroinvertébrés. *Revue des Sciences le l'Eau* **2**: 621-640.

GUEROLD F., VEIN D. & JACQUEMIN G. 1991. Les peuplements d'éphèmèroptères de plécoptères et de trichoptères des ruisseaux acides et non acides du massif vosgien: première approche. *Revue des Sciences le l'Eau* **4**: 299-314.

*GUTHRIE R.K., CHERRY D.S., DAVIS E.M. & MURRAY H.E. 1981. Establishment of biotic communities within a newly constructed ash settling basin and its drainage system. Pp. 243-254. In: *Ecological assessments of effluents impacts on communities of indigenous aquatic organisms.* (Eds Bates J.M. & Weber C.I.). American Society for Testing and Materials STP 730.

HABERER K. & NORMANN S. 1971. Metallspuren im Wasser. *Vom Wasser* **38**: 157-182.

HAIGE E.H. & DAVIES-COLEMAN H.D. 1999. The development of a production facility for standard laboratory test organisms for ecotoxicological research. Water Research Commission Report No. 755/1/99. Water Research Commission, Pretoria, South Africa.

HALL K.J. & ANDERSON B.C. 1988. The toxicity and chemical composition of urban stormwater runoff. *Canadian J. civil Eng.* **15**: 98-106.

HALL L.W. 1987. Acidification effects on larval striped bass, *Morone saxatilis*, in Chesapeake Bay tributaries : a review. *Wat. air & soil Pollut.* **35:** 87-96.

HALL L.W., HALL W.S., BUSHONG S.J. & HERMAN R.L. 1987. *In situ* striped bass (*Morone saxatilis*) contaminant and water quality studies in the Potomac River. *Aquatic Toxicol.* **10**: 73-99.

HALL R.J. 1990. Relative importance of seasonal, short-term pH disturbances during discharge variation on a stream ecosystem. *Can. J. Fish. Aquat. Sci.* **47**: 2261-2274

HALL R.J. & IDE F.P. 1987. Evidence of acidification effects on stream insect communities in central Ontario between 1937 and 1985. *Can. J. Fish. aquat. Sci.* **44**: 1652-1657.

HALL R.J., DRISCOLL C.T.D. & LIKENS G.E. 1987. Importance of H^+ and aluminum in the resulting structure and function of stream ecosystems: an experimental test. *Freshwater. Biol.* **18**: 17-43.

HANSEN S.R. & GARTON R.R. 1982. Ability of standard toxicity tests to predict the effects of the insecticide diflubenzuron on laboratory stream communities. *Can. J. Fish. Aquat. Sci.* **39**: 1273-1288.

HARAMIS G.M. & CHU D.S. 1987. *Acid rain effects on waterfowl: use of Black Duck broods to assess food resources of experimentally acidified wetlands.* ICBP Technical Publication **6**: 173-181.

HARDING J.S., BENFIELD E.F., BOSLTAD P.V., HELFMAN G.S. & JONES E.B.D. 1998. Stream biodiversity: the ghost of land use past. *Proc. Natl. Acad. Sci.* **95**: 14843-14847.

HARRIS G.P. 2001. Biogeochemistry of nitrogen and phosphorus in Australian catchments, rivers and estuaries: effects of land use and flow regulation and comparisons with global patterns. *Mar. Freshwater Res.* **52**: 139-149.

HARRISON A.D. 1958. The effects of sulphuric acid pollution on the biology of streams in the Transvaal. *Verh. Int. Ver. Limnol.* **13**: 603-610.

HARRISON A.D. 1962. Some environmental effects of coal and gold mining on the aquatic biota. *Biological problems in water pollution.* Third seminar-1962.

HARRISON A.D. 1965. Some environmental effects of coal and gold mining on the aquatic biota. Pp 270-274 In: *The relation of land use to the aquatic environment.*

HARRISON A.D., NDUKU W. & HOOPER A.S.C. 1966. The effect of a high magnesium-to-calcium ratio on the egg-laying rate of an aquatic planorbid snail, *Biomphalaria pfeifferi*. *Ann. trop. Med. Paras.* **60**: 212-214.

HARRISON A.R. & ELSWORTH J.F. 1958. Hydrobiological studies on the Great Berg River, Western Cape Province. Part 1. General description, chemical studies and main features of the flora and fauna. *Trans. roy. Soc. S. Afr.* **35**: 125-226.

HARRISON A.R. & FARINA T.D.W. 1965. A naturally turbid water with deleterious effects on the egg capsules of planorbid snails. *Ann. trop. Med. Parasit.* **59**: 327-330.

HART B.T., ANGEHRN-BETTINAZZI C., CAMPBELL I.C. & JONES M.J. 1992. *Australian water quality guidelines.* Australian & New Zealand Environment & Conservation Council.

HART B.T., BAILEY P., EDWARDS R., HORTLE K., JAMES K., McMAHON A., MEREDITH C. & SWADLING K. 1990. Effects of salinity on rivers, streams and wetland ecosystems in Victoria, Australia. *Wat. Res.* **24**: 1103-1117.

HART B.T., BAILEY P., EDWARDS R., HORTLE K., JAMES K., McMAHON A., MEREDITH C. & SWADLING K. 1991. A review of salt sensitivity of the Australian freshwater biota. *Hydrobiol.* **210**: 105-144.

HART C.W. & FULLER S.C.H. 1974. *Pollution ecology of freshwater invertebrates.* Academic Press, New York. 389pp.

HATAKEYAMA S. *et al.* 1988. Population dynamics of three species of chironomids and a mayfly (*Baetis thermicus*) dominant in the Mazawa River highly polluted with heavy metals. *Kokuritsu Kogai Kenkyusho Kenkyu Kokoku* (Jap.) **114**: 177.

HAUG A. 1984. Molecular aspects of aluminum toxicity. *CRC Crit. Rev. plant Sci.* **1**: 345-373.

HAVAS M. & LIKENS G.E. 1985. Changes in Na influx and outflux in *Daphnia magna* as a function of elevated Al concentrations in soft water at low pH. *Proc. Natl. Acad. Sci.* **82**: 7345-7349.

HAVENS K.E. 1990. Aluminium binding to ion exchange sites in acid-sensitive versus acid-tolerant cladocerans. *Environ. Poll.* **64**: 133-141.

HAVENS K.E. & DECOSTA J. 1987. The role of aluminum contamination in determining phytoplankton and zooplankton responses to acidification. *Wat. air & soil Poll.* **33**: 277-293.

HAWKES H.A. 1979. Invertebrates as indicators of river water quality. In: *Biological indicators of water quality*. (Eds. James A. & Evison L.). John Wiley & Sons, Chichester.

HAWKES H.A. 1982. Biological surveillance of rivers. *J. wat. Poll. Contr. Fed.* 329-342.

HAWKES H.A. & DAVIES L.J. 1971. Some effects of organic enrichment on benthic invertebrate communities in stream riffles. In: *The scientific management of animal and plant communities for conservation*. (Eds. Duffey E. & Watts A.S.). Blackwell Scientific Publications, Oxford.

HAWKINS C.P. & NORRIS R.H. 2000. Performance of different landscape classifications for aquatic bioassessments: introduction to the series. *J. N. Am. Benthol. Soc.* **19** (3): 367-369.

HEAL K.-V. 2001. Manganese and land-use in upland catchments in Scotland. *The Science of the Total Environment* 265: 169-179.

HEATH R.G.M. & CLAASSEN M. 1999. *An overview of the pesticide and metal levels present in populations of the larger indigenous fish species of selected South African rivers.* Water Research Commission Report No. 428/1/99. South African Water Research Commission, Pretoria, South Africa.

HELLAWELL J.M. 1986. *Biological indicators of freshwater pollution and environmental management*. Elsevier Applied Science, London. 546pp.

HELSEL D.R., KIM J.I., GRIZZARD T.J., RANDALL C.W. & HOEHN R.C. 1979. Land use influences on metals in storm drainage. *J. wat. Poll. Contr. Fed.* **51**: 709-717.

HENLEY W.F., PATTERSON M.A., NEVES R.J. & LEMLY A.D. 2000. Effects of sedimentation and turbidity on lotic food webs: a concise review for natural resource managers. *Reviews in Fisheries Science* **8**(2): 125-139.

HENRIKSEN A., EKEBIRGE E.J.S., FJELLHEIM A., RADDUM G.G., ROSSELAND B.O. & SKOGHEIM O.K. 1987. *Water chemistry - fish decline: episodic changes in water quality effects on fish and invertebrates*. Report to SWAP, Bergen, Norway.

HERBERT D.W.M. & MERKENS J.C. 1961. The effect of suspended mineral solids on the survival of trout. *Int. J. air wat. Poll.* **5**: 46-55.

HERBERT D.W.M. & RICHARDS J.M. 1963. The growth and survival of fish in some suspensions of solids of industrial origin. *Int. J. air wat. Poll.* **7**: 297-302.

HERBST G.N. & MIENIS H.K. 1985. Aquatic invertebrate distribution in Nahal Tanninim, Israel. *Israel J. Zool.* **33**: 51-62.

HERMANUTZ R.O., HEDTKE S.F., ARTHUR J.W., ANDREW R.W. & ALLEN K.N. 1987. Ammonia effects on macroinvertebrates and fish in outdoor experimental streams. *Environ. Poll.* **47**: 249-283.

HERRMANN J. 1987. Aluminium impact on freshwater invertebrates at low pH: a review. *Earth Sciences* **11**: 157-175.

HERRMANN J., DEGERMAN E., GERHARDT A., JOHANSSON C., LINGDELL P.-E. & MUNIZ I.P. 1993. Acid-stress effects on stream biology. *Ambio* 22: 298-307.

HICKEY C.W. 1989. Sensitivity of four New Zealand cladoceran species and *Daphnia magna* to aquatic toxicants. *N.Z. J. Mar. Freshwat. Res.* **23**: 131-137.

HILL A.R. 1988. Factors influencing nitrate depletion in a rural stream. *Hydrobiol.* **160**: 111-122.

HILSENHOFF W.L. 1988. Rapid field assessment of organic pollution with a family-level biotic index. *J. N. Am. Benthol. Soc.* **7**: 65-68.

HOBSON J.F. & BIRGE W.J. 1989. Acclimation-induced changes in toxicity and induction of metallothionein-like proteins in the fathead minnow following sublethal exposure to zinc. *Environ. Toxicol. Chem.* **8**: 157.

HOFFMAN E.J., LATIMER J.S., MILLS G.L. & QUINN J.G. 1982. Petroleum hydrocarbons in urban runoff from a commercial land use area. *J. wat. Poll. Contr. Fed.* **54**: 1517-1525.

HOGG I.D. & NORRIS R.H. 1991. Effects of runoff from land clearing and urban development on the distribution and abundance of macro-invertebrates in pool areas of a river. *Aust. J. mar. freshwater Res.* **42**: 507-518.

HOGG I.D., WILLIAMS D.D., EADIE J.M. & BUTT S.A. 1995. The consequences of global warming for stream invertebrates: a field simulation. *J. Thermal Biol.* **20**(1-2): 199-206.

HOLCOMBE G.W., BENOIT D.A., LEONARD E.N. & MCKIM J.M. 1976. Long-term effects of lead exposure on three generations of brook trout (*Salvelinus fontinalis*). *J. fish. Res. Bd Can.* **33**: 1731-1741.

HOLCOMBE G.W., PHIPPS G.L., SULAIMAN A.H & HOFFMAN A.D. 1987. Simultaneous multiple species testing: acute toxicity of 13 chemicals to 12 diverse freshwater amphibian, fish, and invertebrate families. *Arch. environ. Contam. Toxicol.* **16**: 697-710.

HORNER R.R. & WELCH E.B. 1982. *Impacts of channel reconstruction in the Pilchuck River.* Washington State Department of Transportation Highway Runoff Water Quality Research Project, University of Washington, Washington, U.S.A. Report No. 15.

HOWELLS G.D. 1983. The effects of power station cooling water discharges on aquatic ecology. *J. wat. Poll. Contr. Fed.*

HRUBY T. 1987. Using similarity measures in benthic impact assessments. *Environ. Monit. Assess.* **8**: 163-180.

HUGHES J.S. 1973. Acute toxicity of thirty chemicals to striped bass (*Morone saxatilis*). *Proc. western Assoc. State Game Fish Comm.* **53**: 399-413.

HUGHES R.M. 1995. Defining acceptable biological status by comparing with reference conditions. In: *Biological assessment and criteria: tools for water resource planning and decision making.* (Eds W.S. Davies & T.P. Simon), pp 31-47. Lewis Publishers, Florida, U.S.A.

HUNTER J.V., SABATINO T., GOMPERTS R. & MCKENZIE M.J. 1979. Contribution of urban runoff to hydrocarbon pollution. *J. wat. Poll. Contr. Fed.* **51**: 2129-2138.

HURLBERT S.H. 1975. Secondary effects of pesticides on aquatic ecosystems. *Residue Reviews* **57**: 81-150.

HUTCHINS F.E. 1979. *The toxicity of pulp and paper mill effluent: a literature review.* United States Environmental Protection Agency Report No. EPA-600/3-79-013.

HUTCHINSON G.E. 1975. *A treatise on limnology.* Vol. I Geography, physics and chemistry. Part 2 Chemistry of Lakes. John Wiley & Sons, Chichester.

HUTCHINSON T.C. & STOKES P.M. 1975. Heavy metal toxicity and algal bioassays. Pp. 320-343. In: *Water Quality Parameters.* American Society for Testing and Materials STP 573.

HYNES H.B.N. 1960. *The biology of polluted waters.* Liverpool University Press, Liverpool. 202pp.

HYNES H.B.N. 1969. The enrichment of streams. In: *Eutrophication: causes, consequences and correctives.* Proceedings of a symposium. National Academy of Sciences, Washington.

ILLIES J. 1964. The invertebrate fauna of the Huallaga, a Peruvian tributary of the Amazon river, from the sources down to Tingo Mario. *Verh. Int. Verein. Limnol.* **15**: 1077-1083.

INGERSOLL C.G. *et al.* 1990. Toxicity of organic and inorganic selenium to *Daphnia magna* (Cladoceran) and *Chironomus riparius* (Diptera). *Environ. Toxicol. Chem.* **9**: 1171.

INGERSOLL C.G., LAPOINT T.W., BRECK J. & BERGMAN H.L. 1985. An early life history stage brook trout (*Salvelinus fontinalis*) bioassay testing the independent and combined effects of pH, calcium and aluminium in low conductivity water. *Air Poll. Acid Rain Rep.* **21**: 42-48. United States Fish and Wildlife Service.

JACOBSEN D. 2000. Gill size of trichopteran larvae and oxygen supply in streams along a 4000-m gradient in altitude. *J. N. Am. Benthol. Soc.* **19**(2): 329-343.

JAFFE R. 1991. Fate of hydrophobic organic pollutants in the aquatic environment: a review. *Environ. Poll.* **69**: 237-257.

JAMES A. & EVISON L. 1979. *Biological indicators of water quality.* John Wiley & Sons, Chichester.

JANA S.R & BANDYOPADHYAYA N. 1987. Effect of heavy metals on some biochemical parameters in the freshwater fish *Channa punctatus*. *Environ. Ecol.* **5**: 488.

JANSEN A. & HEALEY M. 2003. Frog communities and wetland condition: relationships with grazing by domestic livestock along an Australian floodplain river. *Biol. Cons.* **109**: 207-219.

JANSSEN C.R., DE SCHAMPHELAERE K., HEIJERICK D., MUYSSEN B., LOCK K., BOSSUYT B., VANGHELUWE M. & VAN SPRANG P. 2000. Uncertainties in the environmenal risk assessment of metals. *Human and Ecological Risk Assessment* **6**: 1003-1018.

JANSSENS DE BISTHOVEN, L.G., TIMMERMANS K.R. & OLLEVIER F. 1992. The concentration of cadmium, lead, copper and zinc in *Chironomus* gr. *thummi* larvae (Diptera, Chironomidae) with deformed *versus* normal menta. *Hydrobiol.* **239**: 141-149.

JEWASKIEWITZ S.M. & LOMBARD R. 1987. A case study of pollution control measures for a rehabilitated open cast coal mine in Northern Natal. Pp. 323-334. In: *Proceedings of the International Conference on Mining and Industrial Waste Management.* Johannesberg, South Africa.

JOHNSON C.A. & THORNTON I. 1987. Hydrological and chemical factors controlling the concentrations of Cu, Fe, Zn and As in a river system contaminated by acid mine drainage. *Wat. Res.* **21**: 359-365.

JOHNSON M., O'GORMAN K., GALLAGHER M., CLENAGHAN C., GILLER P.S. & O'HALLORAN J. 2000. The identification and amelioration of the impacts of clearfelling operations on lotic macroinvertebrates and salmonids in Ireland. *Verh. Internat. Verein. Limnol.* **27**: 1565-1570.

JOHNSON R. 1998. The forest cycle and low river flows: a review of UK and international studies. *Forest Ecol. Manag.* **109**: 1-7.

*JOHNSON W.E., FENDINGER N.J. & PLIMMER J.R. 1991. Solid-phase extraction of pesticides from water: possible interferences from dissolved organic matter. Pp. 1510-1513. In: *Analytical Chemistry.*

JOHNSON W.W & FINLEY M.T. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. *U.S. Fish Wildl. Serv. Res. Publ.* **137**. 98pp.

JONES G.A., BRIERLEY S.E., GELDENHUIS S.J.J. & HOWARD J.R. 1989. *Research on the contribution of mine dumps to the mineral load in the Vaal barrage.* Water Research Commission report No. 136/1/89, Pretoria, South Africa.

JONES J.G. 1990. Pollution from fish farms. *J. Inst. wat. environ. Manag.* **4**: 14-18.

JONES J.R.E. 1939. The relative toxicity of salts of lead, zinc and copper to the stickleback (*Gasterosteus aculeatus* L) and the effect of calcium on the toxicity of lead and zinc salts. *J. Exp. Biol.* **15**: 394-407.

JONES R.C. & REDFIELD G.W. 1984. Effects of urban stormwater runoff on reservoir phytoplankton. *Verh. Internat. Verein. Limnol.* **22**: 1486-1492.

JOSKA M.A.P., DAY J.A., BOULLE J. & ARCHIBOLD S. In press. *An investigation into the feasibility of using protozoans as indicators of water quality.* Water Research Commission Report for project K5/1017. Water research Commission, Pretoria, South Africa.

KAHL J.S., NORTON S.A., HAINES T.A., ROCHETTE E.A., HEATH R.H. & NODVIN S.C. 1992. Mechanisms of episodic acidification in low-order streams in Maine, USA. *Environ. Poll.* **78**: 37-44.

KAISER M., IRMER U & WEILER K. 1989. Monitoring of water quality: seasonal variations of heavy metals in sediment, suspended particulate matter and tubificids of the Elbe river. *Environ. Tech. Letters* **10**: 845-854.

KARR J.R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* **66**: 21-27.

KARR J.R. 1999. Defining and measuring river health. *Freshwat. Biol.* **41**: 221-234.

KARR J.R. & DUDLEY D.R. 1981. Ecological perspectives on water quality goals. *Environ. Manag.* **1**: 55-68.

KARR J.R., FAUSCH K.D., ARGERMIER P.L., YANT P.R. & SCHLOSSER I.J. 1986. Assessing biological integrity in running waters: a method and its rationale III. *Nat. Hist. Surv. Spec. Publ.* **5**. 28pp.

KAY W.R., HALSE S.A., SCANLON M.D. & SMITH M.J. 2001. Distribution and environmental tolerance of aquatic macroinvertebrate families in the agricultural zone of southwestern Australia. *J. N. Am. Benthol. Soc.* **20**(20): 182-199.

KEENEY W.L., BRECK W.G., VAN LOON G.W. & PAGE J.A. 1976. The determination of trace metals in *Cladophora glomerata* as a potential biological indicator. *Wat. Res.* **10**: 981-984.

KELLY M.G. 1998. Use of community-based indices to monitor eutrophication in rivers. *Environ. Cons.* **25**: 22-29.

KEMP M.J. & DODDS W.K. 2001. Centimeter-scale patterns in dissolved oxygen and nitrification rates in a prairie stream. *J. N. Am. Benthol. Soc.* **20**(3): 347-357.

KEMP P.M. 1967. Hydrological studies on the Tugela River System, Part IV - Acidic drainage from mines in the Natal coalfields. *Hydrobiol.* **29**: 393-425.

KEMPER N.P. (1999). RVI: *Riparian Vegetation Index.* Water Research Commission Report No. K5/850. Water Research Commission, Pretoria, South Africa.

KEMPSTER P.L., HATTINGH W.H.J & VAN VLIET H.R. 1980. *Summarised water quality criteria.* Department of Environmental Affairs Technical Report TR 108. Pretoria, South Africa.

KERR S.R. & VASS W.P. 1973. Pesticide residues in aquatic invertebrates. In: *Environmental pollution by pesticides.* Plenum Press, London.

KHAN A.T., WEIS J.S. & D'ANDREA L. 1988. Studies of cadmium tolerance in two populations of grass shrimp, *Palaemonetes pugio*. *Bull. environ. Contam. Toxicol.* **40**: 30-34.

KHANGAROT B.S. & RAY P.K. 1987. Correlation between heavy metal acute toxicity values in *Daphnia magna* and fish. *Bull. environ. Contam. Toxicol.* **38**: 722-726.

KHANGAROT B.S. & RAY P.K. 1989. Investigation of correlation between physiochemical properties of metals and their toxicity to the water flea *Daphnia magna* Straus. *Ecotoxicol. environ. Saf.* **18**: 109.

KHLEBOVICH V.V. 1990. Some physico-chemical and biological phenomena in the salinity gradient. *Limnologica (Berlin)* **20**: 5-8.

KIFFNEY P.M. & CLEMENTS W.H. 1994a. Structural responses of benthic macroinvertebrate communities from different stream orders to zinc. *Environ Toxicol. Chem.* **13**: 389-395.

KIFFNEY P.M. & CLEMENTS W.H. 1994b. Effects of heavy metals on a macroinvertebrate assemblage from a Rocky Mountain stream in experimental microcosms. *J. N. Am. Benthol. Soc.* **13**: 511-523.

KILANI J.S. 1992. Studies on the treatment of dairy wastes in an algal pond. *Water S.A.* **18**: 57-62.

KIMBOROUGH R.A. & LITKE D.W. 1996. Pesticides in streams draining agricultural nd urban areas in Colorado. *Environ. Sci. Technol.* **30:** 908-916.

KING J.M & THARME R.E. 1994. *Assessment of the instream flow of rivers.* Water Research Commission Report No. 295/1/94. Water Research Commission, Pretoria.

KING J.M. & LOUW D. 1998. Instream flow assessments for regulated rivers in South Africa using the Building Block methodology. *Aquat. Ecosyst. Health Manag.* **1**: 109-124.

KING J.M., BROWN C.A. & SABET H. 2002. A scenario-based holistic approach to environmental flow assessments for regulated rivers. *Rivers Research and Management* – Enviro Flows 2002 Edition.

KING J.M., DAY J.A., HURLY P.R., HENSHALL-HOWARD M-P. & DAVIES B.R. 1988. Macroinvertebrate communities and environment in a southern African mountain stream. *Can. J. Fish. Aquat. Sci.* **45**: 2168-2181.

KING J.M., THARME R.E. & BROWN C.A. 1999. *World Commission on Dams thematic report: definition and implementation on instream flows.* Thematic review for the World Commission on Dams, Cape Town. 60pp.

KING J.M., THARME R.E. & DE VILLIERS M.S. (eds). 2000. *Environmental flow assessments for rivers: manual for the Building Block Methodology.* Water research Commission Technology Transfer Report No. TT131/00. Pretoria, South Africa. 339 pp.

KINVIG R.G. & SAMWAYS M.J. 2000. Conserving dragonflies (Odonata) along streams running through commercial forestry. *Odontologia* **29**(3): 195-208.

KIRK J.T.O. 1985. Effects of suspensoids (turbidity) on penetration of solar radiation in aquatic ecosystems. *Hydrobiol.* **125**: 195-208.

KIRK J.T.O., AKHURST E.G.J., ALLANSON B.R., GROBBELAAR J.U., HELY-HUTCHINSON J.R., MELACK J.M., PATTERSON J.C., ROBARTS R.D. & VINER A.B. 1984. Turbidity and suspensoids. Pp. 108-133. In: *Limnological criteria for management of water quality in the southern hemisphere.* (Eds. Hart R.C. & Allanson B.R.). South Africa National Scientific Programmes Rep. No. 93. Council for Scientific and Industrial Research, Pretoria, South Africa.

KLEIN L.A., LNANG M., NASH N. & KIRSCHNER S.L. 1974. Sources of metal in New York City waste-water. *J. wat. Poll. Contr. Fed.* **46**: 2653-2662.

KLEYNHANS C.J. (1999). The development of a fish index to assess the biological integrity of South African rivers. *Water SA* **25** (3): 265-278.

KLEYNHANS C.J., SCHULZ G.W., ENGELBRECHT J.S. & ROUSSEAU F.J. 1992. The impact of a paper mill effluent spill on the fish populations of the Elands and Crocodile Rivers (Incomati System, Transvaal). *Water S.A.* **18**: 73-80.

*KOLKWITZ R. & MARSSON M. 1908. Ökologie der pflanzlichen Saprobien. *Berichte der Deutschen Botanischen Gesellschaft* **26a**: 505-519.

*KOLKWITZ R. & MARSSON M. 1909. Ökologie der tierschen Saprobien. Beiträge zur Lehre von der biologischen Gewässerbeurteiling. *Int. Rev. Ges. Hydrobiol.* **2**: 126-152.

*KORZENIEWSKI K., BANAT Z. & MOCZULSKA A. 1982. Changes in water of the Uniesc and Skotawa rivers, caused by intensive trout culture. *Polskie Archiwum Hydrobiologii* **29**: 683-691.

*KOVALAK W.P. 1981. Assessment and prediction of impacts of effluents on communities of benthic stream macroinvertebrates. Pp. 255-263. In: *Ecological assessments of effluent impacts on communities of indigenous aquatic organisms.* (Eds Bates J.M. & Weber C.I.). American Society for Testing and Materials STP 730.

KOWNACKI A, DUMNICKA E., GRABACKA E., KAWECKA B. & STARZECKA A. 1985. Stream ecosystems in mountain grassland (West Carpathians) 14. The use of the experimental stream method in evaluating the effect of agricultural pollution. *Acta Hydrobiol* **27**: 381-400.

KRATZ K.W., COOPER S.D. & MELACK J.M. 1994. Effects of single and repeated experimental acid pulses on invertebrates in a high altitude Sierra Nevada stream. *Freshwat. Biol.* **32**: 161-183.

KRISHNAJA A.P. & REGE M.S. 1982. Induction of chromosomal aberrations in fish *Boleophthalmus dussumieri* after exposure *in vivo* to mitomycin C and heavy metals mercury, selenium, and chromium. *Mutat. Res.* **102**: 71-82.

KUHN R., PATTARD M., PERNAK K & WINTER A. 1989. Results of the harmful effects of water pollutants to *Daphnia magna* in the 21-day reproductive test. *Wat. Res.* **23**: 501-510.

LA POINT T.W. & WALLER W.T. 2000. Field assessments in conjunction with whole effluent toxicity testing. *Environ. Toxicol. Chem.* **19**(1): 14-24.

LACKLY M.B. & McARTHUR J.V. 2000. Macroinvertebrate recovery of a post-thermal stream: habitat structure and biotic function. *Ecological Engineering* **15**: S87-S100.

LACROIX G.I. 1992. Mitigation of low stream pH and its effects on salmonids. *Environ. Poll.* **78**: 157-164.

LAMPKIN A.J. & SOMMERFELD M.R. 1986. Impact of mine-acid drainage from abandoned spoils on the chemistry of an intermittent stream from the arid Southwest. *Hydrobiol.* **139**: 135-142.

LANGFORD T.E. & DAFFERN J.R. 1975. The emergence of insects from a British river warmed by power station cooling-water. Part 1. The use and performance of insect emergence traps in a large, spate-river and the effects of various factors on total catches upstream and downstream of the cooling-water outfalls. *Hydrobiol.* **46**: 71-114.

LARSEN R.E., MINER J.R., BUCKHOUSE J.C. & MOORE J.A. 1994. Water-quality benefits of having cattle manure deposited away from streams. *Bioresource Tech.* **48**: 113-118.

LEARNER M.A., WILLIAMS R., HARCUP M. & HUGHES B.D. 1971. A survey of the macrofauna of the River Cynon, a polluted tributary of the River Taff (South Wales). *Freshwater Biol.* **1**: 339-367.

LELAND H.V. & CARTER J.L. 1984. Effects of copper on species composition of periphyton in a Sierra Nevada, California, stream. *Freshwater Biol.* **14**: 281-296.

LELAND H.V. & KENT E. 1981. Effects of copper on microfaunal species composition in a Sierra Nevada, California, stream. *Verh. int. Ver. Limnol.* **21**: 819-829.

LEMKE A.E. 1983. Agrochemical hazards in aquatic systems and evaluation of such hazards with emphasis on developing countries. *Proceedings of the International Conference on environmental hazards from Agrochemicals.* **1:** 417-433.

LEMLY A.D. 1993. Subsurface agricultural irrigation drainage: the need for regulation. *Reg. Toxicol. Pharm.* **17**: 157-180.

LEMLY A.D. 1994. Irrigated agriculture and freshwater wetlands: a struggle for coexistance in the western United States. *Wetlands Ecol. Manag.* **3**(10): 3-15.

LETTERMAN R.D. & MITSCH W.J. 1978. Impact of mine drainage on a mountain stream in Pennsylvania. *Environ. Poll.* **17**: 53-72.

LEVY J., BARRETT A.J. & COOK B.J. 1987. Pollution control for coal waste disposal. pP. 359-365. In: *Proceedings of the International Conference on Mining and Industrial Waste Management.* Johannesburg, South Africa.

LEWIS W.M & MORRIS D.P. 1986. Toxicity of nitrite to fish: a review. *Trans. Am. Fish. Soc.* **115**: 183-195.

LIESS M. & SCHULZ R. 1996. Chronic effects of short-term contamination with the pyrethroid insecticide fenvalerate on the caddisfly *Limnephilus lunatus*. *Hydrobiol.* **324:** 99-106.

LIESS M., SCHULZ R., LIESS M. H.-D., ROTHER B. & KREUZIG R. 1999. Determination of insecticide contamination in agricultural headwater streams. *Wat. Res.* **33:** 239-247.

LIMA, A.R., CURTIS C., HAMMERMEISTER D.E., MARKEE T.P., NORTHCOTT C.E. & BROOKE L.T. 1976. Acute and chronic toxicities to arsenic (III) to fathead minnows, flagfish, daphnids, and an amphipod. *Arch. environ. Contam. Toxicol.* **13**: 595-601.

LIN Y., JING S., LEE D. & WANG T. 2002. Nutrient removal from aquaculture wastewater using a constructed wetlands system. *Aquaculture* **209**: 169-184.

LISTON P. & MAHER W. 1986. Trace metal export in urban runoff and its biological significance. *Bull. environ. Contam. Toxicol.* **36**: 900-905.

LITAOR M.I. 1987. Aluminum chemistry: fractionation, speciation, and mineral equilibria of soil interstitial waters of an alpine watershed, Front Range, Colorado. *Geochim. et Cosmochim. Acta* **51**: 1285-1295.

LIVINGSTON R.J. 1977. Review of current literature concerning the acute and chronic effects of pesticides on aquatic organisms. *CRC crit. Rev. environ. Control* **7**: 325-351.

LLOYD D.S., KOENINGS J.P. & LAPERRIERE J.D. 1987. Effects of turbidity in freshwaters of Alaska. *North American J. Fish. Manag.* **7**: 18-33.

LLOYD R. & SWIFT D.J. 1976. Some physiological responses by freshwater fish to low dissolved oxygen, high carbon dioxide, ammonia and phenol with particular reference to water balance. Pp. 47-69. In: *The effects of pollutants on aquatic organisms*. (Ed. Lockwood A.P.M.), Cambridge University Press, Cambridge.

LOCH D.D., WEST J.L. & PERLMUTTER D.G. 1996. The effect of trout farm effluent on the taxa richness of benthic macroinvertebrates. *Aquaculture* **147**: 37-55.

LONDON L., DALVIE M.A., CAIRNCROSS E. & SOLOMONS A. 2000. *The quality of surface and groundwater in the rural Western Cape with regard to pesticides*. Water Research Commission Report No. 795/1/00. South African Water Research Commission, Pretoria, South Africa.

LOOS M.A., BOSCH C. & MARE J. 1990a. *Research on the inhibition of bacterial oxidation of pyrite and the concommitant acid mine drainage: Part 2. Investigations on coal waste dumps.* Water Research Commission Report No. 132/2/90, Pretoria, South Africa.

LOOS M.A., CONRADIE J.M., WHILLIER P.A., MARE J. & BOSCH C. 1990b. *Research on the inhibition of bacterial oxidation of pyrite and the concommitant acid mine drainage: Part 1. Investigations on gold mine sand dumps.* Water Research Commission Report No. 132/1/90, Pretoria, South Africa.

LOWRANCE R.R, LEONARD R. & SHERIDAN J. 1985. Managing riparian ecosystems to control nonpoint pollution. *J. Wat. Soil. Cons.* **40**: 87-91.

LUEDTKE R.J., BRUSVEN M.A. & WATTS F.J. 1976. Benthic insect community changes in relation to in-stream alterations of a sediment-polluted stream. *Melanderia* **23**: 21-39.

LUGER M. & DAVIES B.R. 1993. Environmentally insensitive design and construction: the Liesbeek River walkway, Cape Town. *Environmental Planning and Management* **5** (1): 4-11.

MAAGE A. 1990. Comparison of cadmium concentrations in Atlantic Salmon (*Salmo salar*) fry fed different commercial feeds. *Bull. environ. Contam. Toxicol.* **44**: 770.

MACKERETH F.J.H., HERON J. & TALLING J.F. 1978. Water analysis: some revised methods for limnologists. *Sci. Public. Freshw. Biol. Assoc.* **36**: 1-120.

MACRAE T.H. & PANDEY A.S. 1991. Effects of metals on early life stages of the brine shrimp, *Artemia*: a developmental toxicity assay. *Arch. environ. Contam. Toxicol.* **20**: 247-252.

MAGDYCH W.P. 1984. Salinity stress alonga complex river continuum: effects on mayfly (Ephemeroptera) distributions. *Ecology* **65**: 1662-1672.

MALAN H.L. & DAY J.A. 2000a. *Linking discharge, water quality and biotic response in rivers: a literature review.* Water Research Commission Report No. 956/2/02. Water Research Commission, Pretoria, South Africa.

MALAN H.L. & DAY J.A. 2000b. *Development of numerical methods for predicting relationships between stream flow, water quality and biotic responses in rivers.* Water Research Commission Report No. 956/1/02. Water Research Commission, Pretoria, South Africa.

MALAN H.L. & DAY J.A. In press. *Linking flows, water quality and potential effects on aquatic biota within the reserve determination process.* Water SA.

MALONEY R.F. 1995. Effect of the herbicide triclopyr on the abundance and species composition of benthic macroinvertebrates in the Ahuriri River, New Zealand. *N. Z. J. Mar. Freshwat. Res.* **29**: 505-515.

MALYAREVSKAYA A.Y. & KARASINA F.M. 1987. Effect of lead nitrate on physiological and biochemical characteristics of certain aquatic invertebrates. *Hydrobiol. J.* **23**: 84-89.

MANCE G. 1986. Water quality standards in relation to the European Community. *J. Inst. wat. Poll. Contr.* **85**: 25-33.

MANN K.H. 1965. Heated effluents and their effects on the invertebrate fauna of rivers. *Proc. Soc. wat. Treatm. & Exam.* **14**: 45-53.

MARCHANT R. 1988. Seasonal and longitudinal patterns in the macroinvertebrate communities of cobbles from the upper La Trobe River, Victoria, Australia. *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie* **23**: 1389-1393.

*MARGALEF R. 1951. Diversidad de especies en las comunidades naturales. *Publnes. Inst. Biol. apl., Barcelona* **6**: 59-72.

MARLIER G., BOUILLON J., DUBOIS T. & LELEUP N. 1955. Le Lac Lungwe. *Acad. Roy. Sci. Col. Bull. Sci.* **1**: 665-676.

MARSALEK J., ROCHFORT Q., GRAPENTINE L., BROWNLEE B. & SALDANHA MATOS J. 2002. Assessment of stormwater impacts on an urban stream with a detention pond. *Wat. Sci. Tech.* **45** (3): 255-263.

MATSCHULLAT J., ANDRAEA H., LESSMANN D., MALESSA V. & SIEWERS U. 1992. Catchment acidification - from the top down. *Environ. Poll.* **77**: 143-150.

MATTHIESSEN P. & LOGAN J.W.M. 1984. Low concentration effects of endosulfan insecticide on reproductive behaviour in the tropical cichlid fish *Sarotherodon mossambicus*. *Bull. Environ. Contamin. Toxicol.* **33:** 575-583.

MAURICE C.G., LOWE R.L., BURTON T.M. & STANFORD R.M. 1987. Biomass and compositional changes in the periphytic community of an artificial stream in response to lowered pH. *Wat. air & soil Poll.* **33**: 165-177.

MAYES M.A., ALEXANDER H.C., HOPKINS D.L. & LATVAITIS P.B. 1986. Acute and chronic toxicity of ammonia to freshwater fish: a site-specific study. *Environ. Toxicol. Chem.* **5**: 437-442.

McCAHON C.P. & POULTON M.J. 1991. Lethal and sub-lethal effects of acid, aluminium and lime on *Gammarus pulex* during repeated simulated episodes in a Welsh stream. *Freshwater. Biol.* **25**: 169-178.

McCOLL R.H.S. 1974. Self-purification of small freshwater streams: phosphate, nitrate and ammonia removal. *N. Z. J. Mar. Freshwater Res.* **8**: 375-388.

McCOLL R.H.S. & HUGHES H.R. 1981. The effects of land use on water quality - a review. *Water & Soil Miscellaneous Publication No. 23.*, Ministry of Works and Development, Wellington, New Zealand. pp. 3-59.

McCONNELL R.P. 1977. Toxicity of molybdenum to rainbow trout under laboratory conditions. In: *Molybdenum in the environment. 2. The geochemistry, cycling, and industrial uses of molybdenum.* Marcel Dekker, New York.

McCORMICK F.H., HILL B.H., PARRISH L.P. & WILLINGHAM W.T. 1994. Mining impacts of fish assemblages in the Eagle and Arkansas rivers, Colorado. *J. Freshwat. Ecol.* **9**: 175-179.

McKEE J.E. & WOLF H.W. 1963. *Water quality criteria.* The resource Agency of California. 2nd Ed. State Water Quality Control Bd., Publ. 3A. 584pp.

McKEE J.E. & WOLF H.W. 1971. *Water quality criteria.* The Resource Agency of California. 2nd Ed. State Water Quality Control Bd., Publ. 3A. 548pp.

McKNIGHT D.M. & BENCALA K.E. 1990. The chemistry of iron, aluminium, and dissolved organic material in three acidic, metal-enriched, mountain streams, as controlled by watershed and in-stream processes. *Wat. Resources Res.* **26** (12): 3087-3100.

McLEAN R.O. & JONES A.K. 1975. Studies of tolerance to heavy metals in the flora of the rivers Ystwyth and Clarach, Wales. *Freshwater Biol.* **5**: 431-444.

McNEILL A. 1989. The effects of a timber preservative spillage on the ecology of the River Lossie. *J. Inst. wat. Poll. Contr.* **3**: 496-504.

McNAUGHTON D.C. 1988. Pesticides and ground water quality; a field study in western Canada. *Agricultural Chemicals and the Environment*: Proceedings of a meeting, Western Branch, Manitoba Institute of Agrologists, 24th February 1988, Brandon, Manitoba.

MEHRLE P.M., BUCKLER D., FINGER S. & LUDKE L. 1984. *Impact of contaminants on striped bass.* Interim Report, U.S. Fish. Wildl. Ser., Columbia National Fisheries Research Laboratory, Columbia, MO, 28pp.

MEINTJIES E., VAN DER MERWE L. & DU PREEZ J.L. 2000. *Qualitative and quantitative evaluation of estrogen and estrogen-mimicking substances in the water environment.* Water Research Commission Report No. 742/1/00. South African Water Research Commission, Pretoria, South Africa.

MENZIE C.A. 1980. Potential significance of insects in the removal of contaminants from aquatic systems. *Wat. air & soil Poll.* **13**: 473-479.

MERRIMAN J.C., ANTHONY D.H.J., KRAFT J.A. & WILKINSON R.J. 1991. Rainy river water quality in the vicinity of bleached kraft mills. *Chemosphere* **23**: 1605-1615.

METCALFE-SMITH J.L. 1991. Biological water quality assessment of rivers based on macroinvertebrate communities. Chapter 3.In: *Rivers handbook.* Monitoring Programmes National Water Research Institute Contribution No. 91-71. Canada.

METSI CONSULTANTS. 2000. Consulting services for the establishment and monitoring of the instream flow requirements for river courses downstream of LHWP dams. *Final Report: Summary of main findings.* Report No. 648-F-02. Authors: King J., Sabet H., Brown C. & Hirst S. Lesotho Highlands Development Authority Contract 648, Maseru, Lesotho.

MEYER J.L. 1997. Stream health: incorporating the human dimension to advance stream ecology. *J. N. Am. Benthol. Soc.* **16** (2): 439-447.

MEYER J.L & LIKENS G.E. 1979. Transport and transformation of phosphorus in a forest stream ecosystem. *Ecology* **60**: 1255-1269.

MHATRE G.N., CHAPHEKAR S.B., RAMANO RAO I.V., PATIL M.R. & HALDAR B.C. 1980. Effect of industrial pollution on the Kalu river system. *Environ. Poll.* **23**: 67-78.

MIGLIORE L. & DE NICOLA GIUDICI M. 1990. Toxicity of heavy metals to *Asellus asquaticus* (L.) (Crustacea, Isopoda). *Hydrobiol.* **203**: 155-164.

MINNS C.K. 1995. Approaches for assessing and managing cumulative ecosystem change, with the Bay of Quinte as a case study: an assay. *J. Aq. Ecosys. Health* **4**: 1-24.

MONSERRAT J.E., RODRIGUEZ E.M. & LOMBARDO R.J. 1991. Effects of salinity on the toxicity of parathion to the estuarine crab *Chasmagnathus granulata* (Decapoda; Grapsidae). *Bull. environ. Contam. Toxicol.* **46**: 569-575.

MOON T.C. & LUCOSTIC C.M. 1979. Effects of acid mine drainage on a southwestern Pennsylvania stream. *Wat. air & soil Poll.* **11**: 377-390.

MOORE J.W. 1980. Seasonal and species-dependent variability in the biological impact of mine wastes in an alpine river. *Bull. Environ. Contam. Toxicol.* **25**: 524-529.

MORGAN M.D. 1987. Impact of nutrient enrichment and alkalinization on periphyton communities in the New Jersey Pine Barrens. *Hydrobiol.* **144**: 233-241.

MORRISSY N.M. 1974. Reversed longitudinal salinity profile of a major river in the south-west of Western Australia. *Aust. J. mar. freshwater Res.* **25**: 327-335.

MOSS B.R. 1980. *Ecology of fresh waters.* Blackwell Scientific Publishers, Oxford.

MOUNT D.I. & STEPHAN C.E. 1969. Chronic toxicity of copper to the fathead minnow (*Pimephales promelas*) in soft water. *J. Fish. Res. Bd. Can.* **26**: 2449-2457.

MULHOLLAND P.J., NEWBOLD D., ELWOOD J.W. & FERREN L.A. 1985. Phosphorus spiralling in a woodland stream: seasonal variations. *Ecology* **66**: 1012-1023.

MULLHOLAND P.J., ELWOOD J.W., PALUMBO A.V. & STEVENSON R.J. 1986. Effect of stream acidification on periphyton composition, chlorophyl, and productivity. *Can. J. Fish. Aquat. Sci.* **43**: 1846-1858.

MUNCY R.J., ATCHISON G.J., BULKLEY R.V., MENZEL B.W., PERRY L.G. & SUMMERFLET R.C. 1979. *Effects of suspended solids and sediment on reproduction and early life of warmwater fishes.* EPA Report 600/3-79-042, United States Environmental Protection Agency, Washington, DC.

MUNIZ I.P. & LEIVESTAD H. 1980. Toxic effects of aluminum on the brown trout *Salmo trutta* L. Pp. 268-269. In: *Ecological impacts of acid rain.* (Eds: Drablos D. & Tollan A.). SNSF Project, Oslo.

MURPHY C.B. & CARLEO P.J. 1978. The contribution of mercury and chlorinated organics from urban runoff. *Wat. Res.* **12**: 531-533.

MURTI R., OMKAR O. & SHUKLA G.S. 1983. Chromium toxicity to a freshwater prawn *Macrobrachium lamarrei* (H.M.-Edwards). *Toxicol. Lett.* **18**: 257-261.

MUSHAK P. 1985. Potential impact of acid precipitation on arsenic and selenium. *Environ. health Perspec.* **63**: 169-180.

MUSIBONO D.E & DAY J.A. 1999. The effect of Mn on mortality and growth in the freshwater amphipod *Paramelita nigroculus* (Barnard) exposed to a mixture of Al and Cu in acidic waters. *Water Research* **33**: 207-213.

NAIMAN R.J. 1976. Primary production, standing stock and export of organic matter in a Mohave Desert thermal stream. *Limnol. Oceanogr.* **21**: 60-73.

NAKANE K., KANG H-S., HONG S-U. & YIM Y-J. 1981. Dynamics of nutrients and heavy metals in the Han River and its basin in the Korean Peninsula. I: Dynamics of nutrients (Na, K, Mg, Ca & P). *Verh. int. Verein. Limnol.* **21**: 886-893.

NATHER KHAN I.S.A. 1991. Effect of urban and industrial wastes on species diversity of the diatom community in a tropical river, Malaysia. *Hydrobiol.* **224**: 175-184.

NATSURV 10 1989. *Water and waste-water management in the tanning and leather finishing industry.* Water Research Commission Project No. 145, TT 44/90. Pretoria, South Africa.

NATSURV 11 1990. *Water and waste-water management in the sugar industry.* Water Research Commission Project No. 145, TT 47/90. Pretoria, South Africa.

NATSURV 12 1990. *Water and waste-water management in the paper and pulp industry.* Water Research Commission Project No. 145, TT49/90. Pretoria, South Africa.

NATSURV 4. 1989. *Water and waste-water management in the dairy industry.* Water Research Commission Project No. 145, TT 38/89. Pretoria, South Africa.

NATSURV 7. 1989. *Water and waste-water management in the red meat industry.* Water Research Commission Project No. 145, TT 41/89. Pretoria, South Africa.

NATSURV 9. 1989. *Water and waste-water management in the poultry industry.* Water Research Commission Project No. 145, TT 43/89. Pretoria, South Africa.

NATURE CONSERVANCY COUNCIL (SCOTLAND). 1990. *Fish farming and the Scottish freshwater environment.* NCC. Contract No.HF3-03-450.

NEAL C. 2001. Alkalinity measurements within natural waters: towards a standardised approach. *The Science of the Total Environment* **265**: 99-113.

NEBEKER A.V. 1972. Effect of low oxygen concentration on survival and emergence of aquatic insects. *Trans. Amer. Fish. Soc.* **4**: 675-679.

NEHRING R.B. 1976. Aquatic insects as biological monitors of heavy metal pollution. *Bull. environ. Contam. Toxicol.* **15**: 147-154.

NEHRING R.B., NISSON R. & MINASIAN G. 1979. Reliability of aquatic insects versus water samples as measures of aquatic lead pollution. *Bull. environ. Contam. Toxicol.* **22**: 103-108.

NELSON S.M. & CAMPBELL S.G. 1995. Integrated assessment of metals contamination in a lotic system using water chemistry, transplanted bryophytes, and macroinvertebrates. *J. Freshw. Ecol.* **10**: 409-420.

NEUMANN M. & DUDGEON D. 2002. The impact of agricultural runoff on stream benthos in Hong King, China. *Wat. Res.* **36**: 3103-3109.

NEWBOLD J.D., ERMAN D.C. & ROBY K.B. 1980. Effects of logging on macroinvertebrates in streams with and without buffer strips. *Can. J. Fish. aquat. Sci.* **37**: 1076-1085.

NISBET T.R. 2001. The role of forest management in controlling diffuse pollution in UK forestry. *Forest Ecol. Manag.* **143**: 215-226.

NIWR 1985. National Institute for Water Research, CSIR. *The limnology of the Hartebeespoort Dam.* South African National Scientific Programmes Report No. 110. Council for Scientific and Industrial Research, Pretoria. 269pp.

NOEL D.S., MARTIN C.W. & FEDERER C.A. 1986. Effects of forest clearcutting in New England on stream macroinvertebrates and periphyton. *Environ. Manag.* **10**: 661-670.

NORDLIE K.J. & ARTHUR J.W. 1981. Effect of elevated water temperature on insect emergence in outdoor experimental channels. *Environ. Poll.* (Series A) **25**: 53-65.

NORRIS R.H., LAKE P.S. & SWAIN R. 1982. Ecological effects of mine effluents on the South Esk River, north-eastern Tasmania. III. Benthic macroinvertebrates. *Australian Journal of Mar. Freshwat. Res.* **33**: 780-809.

*NORRSTROM H. 1976. Chemical pulping. *Pure appl. Chem.* **45**: 181-186.

NOVOTNY V. & CHESTES G. 1981. Pollution from impervious urban areas. Pp. 312-347. In: *Handbook of non-point pollution: sources and engineering management.* Van Nostrand Reinhold Environmental Engineering Series, New York.

*NRC (National Research Council). 1992. *Restoration of aquatic ecosystems: science technology and public policy.* National Academic Press, Washington, D.C. USA.

NRCC. 1978. Effects of arsenic in the Canadian environment. *Natl. Res. Coun. Can. Publ. No. NRCC 15391.* 349pp.

NUTTALL P.M. & BIELBY G.H. 1973. The effects of china-clay wastes on stream invertebrates. *Environ. Pollut.* **5**: 77-86.

NWQMS (National Water Quality Management Strategy). 2000. *Australian and New Zealand guidelines for fresh and marine water quality. Volume 2, Aquatic Ecosystems.* Australian and New Zealand Environment and Conservation Council, Agriculture and Resource Management Council of Australia and New Zealand.

NYBERG D. 1974. Breeding systems and resistance to environmental stress in ciliates. *Evolution* **28**: 367-380.

NYBERG D. 1975. Genetic analysis of copper resistance in *Paramecium aurelia* Syn gen 4. *Genetics* **80**: 463-473.

O'KEEFE J.H. & DE MOOR F.C. 1988. Changes in the physico-chemistry and benthic invertebrates of the Great Fish River, South Africa, following an inter-basin transfer of water. *Reg. Rivers Res. Manag.* **2**: 39-55.

O'KEEFFE J.H., BYREN B.A., DAVIES B.R. & PALMER R.W. 1990. The effects of impoundment on the physico-chemistry of two southern African river systems. *Reg. Rivers Res. Manag.* **5**: 97-110.

OCCHIOGROSSO T.J., WALLER W.T. & LAUER G.J. 1979. Effects of heavy metals on benthic macroinvertebrate densities in Foundry Cove on the Hudson River. *Bull. environ. Contam. Toxicol.* **22**: 230-237.

O'CONNOR J.T. 1974. Removal of trace inorganic constituents by conventional water treatment processes. Pp. 99-110. In: *Proceedings of the 16th Water Quality Conference on trace metals in water supplies: occurence, significance, and control.* Univ.Ill., Coll. Eng.

ODENKIRCHEN E.W. & EISLER R. 1988. Chlorpyrifos hazards to fish, wildlife, and invertebrates: a synoptic review. Contaminant Hazard Reviews, Report no. 13: *U.S. Fish. Wildl. Serv. Biol. Rep.* **85** (1.13): 1-34.

ODUM E.P. 1971. *Fundamentals of Ecology.* W.B. Saunders & Co., Philadelphia.

OGBEIBU A.E. & VICTOR R. 1989. The effects of bridge construction on the bank-root macrobenthic invertebrates of a southern Nigerian stream. *Environ. Poll.* **56**: 85-100.

OHLENDORF H.M., CUSTER T.W., LOWE R.W., RIGNEY M. & CROMARTIE E. 1988. Organochlorines and mercury in eggs of coastal Terns and Herons in California, USA. *Colonial Waterbirds* **11**: 85-94.

O'KEEFFE J.H., UYS M. & BRUTON M.N. 1992. Freshwater ecosystems. Pp. 277-315. In: *Environmental management in South Africa* (Eds. Fuggle R.E. & Rabie M.A.). Juta & Co. Ltd., Wetton.

OKORONKWO N. & ODEYEMI O. 1985. Effects of a sewage lagoon effluent on the water quality of the receiving stream. *Environ. Poll.* **37**: 71-86.

*OKOYE B.C.O. 1991. Heavy metals and organisms in the Lagos Lagoon. *Int. J. environ. Studies* **37**: 285-292.

OLADIMEJI A.A. & OFFEM B.O. 1989. Toxicity of lead to *Clarius lazera, Oreochromis nitoticus, Chironomus tentans* and *Benacus* sp. *Wat. air & soil Poll.* **44**: 191.

OLIFF W.D. 1963. Hydrobiological studies on the Tugela River System. Part III: The Buffalo River. *Hydrobiol.* **21**: 355-370.

OLSON T.A. & RUEGER M.E. 1968. Relationship of oxygen requirements to index-organism classification of immature aquatic insects. *Water Poll. Contrl. Fed.* **40**(5): R188-R202.

ORMEROD S.J., BOOLE P., MCCAHON C.P., WEATHERLEY N.S., PASCOE D. & EDWARDS R.W. 1987. Short-term experimental acidification of a Welsh stream: comparing the biological effects of hydrogen ions and aluminium. *Freshwater. Biol.* **17**: 341-356.

OSBORNE L.L. & DAVIES R.W. 1987. The effects of a chlorinated discharge and a thermal outfall on the structure and composition of the aquatic macroinvertebrate communities in the Sheep River, Alberta, Canada. *Wat. Res.* **8**: 913-921.

OSBORNE L.L. & KOVACIC D.A. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. *Fresh. Biol.* **29**: 243-258.

PALMER C.G. 1991. *Benthic assemblage structure, and feeding biology, of sixteen macroinvertebrate taxa from the Buffalo River, Eastern Cape, South Africa.* Unpublished Ph.D Thesis, Rhodes University, South Africa.

PALMER C.G. &SCHERMAN P.A. In press. *Application of an artifical stream system to investigate the water quality tolerances of indigenous, South Africa, riverine macroinvertebrates.* Water Research Commission Report No. 686/--. Water Research Commission, Pretoria, South Africa.

PALMER C.G., GOETSCH P.A. & O'KEEFFE J.H. 1996. Development of a recirculating artificial stream system to investigate macroinvertebrates as water quality indicators. Report No. 475/1/96. Water Research Commission, Pretoria, South Africa.

PALMER C.G., O'KEEFFE J.H. & PALMER A.R. 1991. Are macroinvertebrate assemblages in the Buffalo River, southern Africa, associated with particular biotopes? *J. N. Am. Benthol. Soc.* **10** (4): 349-357.

PAQUIN P.R., SANTORE R.C., WU K.B., KAVVADAS D. & DI TORO D. M. 2000. The biotic ligand model of the acute toxicity of metals to aquatic life. *Environmental Science and Policy.* **3**: S175-S182.

PASCOE D. & MATTEY D.L. 1977. Studies on the toxicity of cadmium to the three-spined stickleback, *Gasterosteus aculeatus*. *J. Fish. Biol.* **11**: 207-215.

PEDERSEN E.R. & PERKINS M.A. 1986. The use of benthic invertebrate fauna for evaluating impacts of urban runoff. *Hydrobiol.* **139**: 13-22.

PERRIN C.J., SHORTREED K.S. & STOCKNER J.G. 1984. An integration of forest and lake fertilisation: transport and transformations of fertiliser element. *Can. J. Fish. Aquatic Sci.* **41**: 253-262.

PETERSON B.J., HOBBIE J.E., HERSHEY A.E., LOCK M.A., FORD T.E., VESTAL J.R., McKINLY V.L., HULLAR M.A.J., MILLER M.C., VENTULLO R.M. & VOLK G.S. 1985. Transformation of a tundra river from heterotrophy to autotrophy by addition of phosphorus. *American Association for the Advancement of Science*: **229**: 1383-1386.

PETERSON M.J., SOUTHWORTH G.R. & CRUMBY W.D. 1996. Monitoring mercury in fish in a stream system receiving multiple industrial inputs. *Environ. Monit. Assess.* **40**: 91-105.

PETITJEAN M.O.G. & DAVIES B.R. 1988. Ecological impacts of inter-basin water transfers: Some case studies, research requirements and assessment procedures in southern Africa. *S. Afr. J. Sci.* **84**: 819-828.

PETRY J., SOULSBY C., MALCOLM I.A. & YOUNGSON A.F. 2002. Hydrological controls on nutrient concentration and fluxes in agricultural catchments. *The Sci. of the Tot. Env.* 294: 95-110.

PETTICREW E.K. 1996. Sediment aggregation and transport in northern interior British Colombia streams. *Int. Ass. Hydrol. Sci. Publ.* **236**: 313-319.

PHIPPS G.L. & HOLCOMBE G.W. 1985. A method for aquatic multiple species toxicant testing: acute toxicity of 10 chemicals to 5 vertebrates and 2 invertebrates. *Environ. Poll.* **38**: 141-157.

PHIPPS G.L., MATTSON V.R. & ANKLEY G.T. 1995. Relative sensitivity of three freshwater benthic macroinvertebrates to ten contaminants. *Arch. Environ. Contam. Toxicol.* **28:** 281-286.

PICKERING Q., CARLE D.O., PILLI A., WILLINGHAM T. & LAZORCHAK J.M. 1989. Effects of pollution on freshwater organisms. *J. wat. Poll. Contr. Fed.* **61**: 998-1042.

PICKERING Q.H. & HENDERSON C. 1966. The acute toxicity of some heavy metals to different species of warm water fishes. *Int. J. air wat. Poll.* **10**: 453-463.

PILLI A., CARLE D., KLINE E., PICKERING Q & LAZORCHAK J. 1988. Effects of pollution on freshwater organisms. *J. wat. Poll. Contr. Fed.* **60**: 994-1065.

PILLSBURY A.F. 1981. The salinity of rivers. *Scientific American* **245**: 32-43.

PIMENTEL R. & BULKLEY R.V. 1983. Concentrations of TDS preferred or avoided by endangered Colorado River fishes. *Trans. Am. Fish Soc.* **112**: 595-600.

PITCHFORD R.J. & VISSER P.S. 1975. The effect of large dams on river water temperature below dams, with special reference to bilharzia and the Verwoerd Dam. *S. Afr. J. Sci.* **71**: 212-213.

PLAFKIN J.L., BARBOUR M.T., PORTER K.D. *et al.* 1989. *Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrate and fish.* United States Environmental Protection Agency Report No. EPA/444/4-89-001. 162pp.

PLAPP Jr. F.W. 1981. The nature, modes of action, and toxicity of insecticides. In: *The handbook of pest management in agriculture.* (Ed. Pimental D.). Vol. 3. CRC Press, Cleveland.

PLÉNET S. 1995. Freshwater amphipods as biomonitors of metal pollution in surface and interstitial aquatic environments. *Freshwat. Biol.* **33**: 127-137.

PONTASCH K.W., NIEDERLEHNER B.R. & CAIRNS J.JR. 1989. Comparisons of single-species, microcosm and field responses to a complex effluent. *Environ. Toxicol. Chem.* **8**: 521-532.

PORTER K.S. 1975. *Nitrogen and phosphorus: Food production, waste and the environment.* Anne Arbor Science Publishers Inc., Michigan, United States. 372pp.

PRINSLOO J.F. & PIETERSE A.J.H. 1994. Preliminary observations on the effect of increased concentrations of totl dissolved salts on growth and photosynthetic rates in different algal species. *Water SA* **20:** 219-222.

PRITCHARD P.H. 1986. Fate of Pollutants. *J. wat. Poll. Contr. Fed.* **58**: 635-644.

QUINN J.M. & HICKEY C.W. 1990. Characterisation and classification of benthic invertebrates communities in 88 New Zealand rivers in relation to environmental factors. *N. Z. J. Mar. Freshwat. Res.* **24**: 387-409.

QUINN J.M., COOPER A.B., DAVIES-COLLEY R.J., RUTHERFORD J.C. & WILLIAMSON R.B. 1997. Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams. *N. Z. J. Mar. Freshwat. Res* **31** (5): 579-597.

QUINN J.M., DAVIES-COLLEY R.J. & HICKEY G.W. 1991. *Inert suspended sediment effects on river ecosystems.* Conservation Science Centre, Department of Conservation, New Zealand. Consultancy Rep. No. 6024/2. 16pp.

QUINN J.M., STEELE G.L., HICKEY C.W. & VICKERS M.L. 1994. Upper thermal tolerances of twelve New Zealand stream invertebrate species. *N. Z. J. Mar. Freshwat. Res.* **28**: 391-397.

RACTLIFFE G. 1991. *The effects of suspended sediments on the macroinvertebrate community structure of a river ecosystem.* Unpublished Honours Project, University of Cape Town, South Africa. 57pp.

RADWAN S., KOWALCZYK C.Z., JARZYNA B., PALEOLOG A., ZWOLSKI W. & KOWALIK W. 1991. Coal mine waters and their influence on the purity ecological state of river and the fish production. *Mar. Poll. Bull.* **23**: 733-736.

RAMELOU G.J., MAPLES R.S., THOMPSON R.L., MUELLER C.S. & WEBER C. 1987. Periphyton as monitors for heavy metal pollution in the Calcasieu River Estuary. *Environ. Poll.* **43**: 247-261.

RANKIN E.T. 1991. The use of biocriteria in the assessment of nonpoint and habitat impacts in warmwater streams. Ohio Environmental Protection Agency.

RAPPORT D.J., COSTANZA R. & McMICHAEL A.J. (1998). Assessing ecosystem health. *Trends in ecology and evolution* **13**: 397-402.

RAUBENHEIMER C.M. & DAY J.A. *1991. In vitro* leaching of two sclerophyllous fynbos plants. *Hydrobiol.* **224**: 167-174.

RAVEN P.J. & GEORGE J.J. 1989. Recovery by riffle macroinvertebrates in a river after a major accidental spillage of chlorpyrifos. *Environ. Poll.* **59**: 55-70.

REHWOLDT R., LASKE L., SHAW C. & WIRHOWSKY E. 1973. The acute toxicity of some heavy metal ions towards benthic organisms. *Bull. environ. Contam. Toxicol.* **10**: 291-294.

REID G.K. & WOOD R.D. 1976. *Ecology of inland water and estuaries.* D. Van Nostrand Company, New York.

REIMER P. & DUTHIE H.C. 1993. Concentrations of zinc and chromium in aquatic macrophytes from the Sudbury and Muskoka regions of Ontario, Canada. *Environ. Poll.* **79**: 261-265.

REVELLE C. & REVELLE P. 1974. *Sourcebook on the environment: the scientific perspective.* Houghton Mifflin Company, Boston.

REYNOLDSON T.B. & METCALFE-SMITH J.L. 1992. *An overview of the assessment of aquatic ecosystem health using benthic macroinvertebrates.* Draft document of Environment Canada, National Water Research Institute.

REYNOLDSON T.B. & WRIGHT J.F. 2000. The reference condition: problems and solutions. In: *Assessing the biological quality of fresh waters: RIVPACS and other techniques* (Eds. Wright J.F., Sutcliffe D.W. & Furse M.T.), Freshwater Biological Association, United Kingdom.

REYNOLDSON T.B., NORRIS R.H., RESH V.H., DAY K.E. & ROSENBERG D.M. (1997). The reference condition: a comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. *J. N. Am. Benthol. Soc.* **16** (4): 833-852.

RICHARDS C., HOST G.H. & ARTHUR J.W. 1993. Identification of predominant environmental factors structuring stream macroinvertebrate communities within a large agricultural catchment. *Freshwater Biology* ***29**: 285-294.*

RICHARDSON J., BOUBEE J.A. & WEST D.W. 1994. Thermal tolerance and preference of some native New Zealand fish. *N. Z. J. Mar. Freshwat. Res.* **28**: 399-407.

RICHEY J. 1982. *Effects of urbanisation on a lowland stream in Western Washington.* Doctoral Dissertation, University of Washington. 248pp.

RICO M.C., HERNANDEZ L.M., GONZALAZ M.J., FERNANDEZ M.A. & MONTERO M.C. 1987. Organochlorine and metal pollution in aquatic organisms sampled in the Donana National Park during the period 1983-1986. *Bull. environ. Contam. Toxicol.* **39**: 1076-1083.

RIMER A.E., NISSEN J.A. & REYNOLDS D.E. 1978. Characterization and impact of stormwater runoff from various land cover types. *J. wat. Poll. Contr. Fed.* **50**: 252-264.

ROBINSON J. 1973. Dynamics of pesticide residues in the environment. In: *Environmental pollution by pesticides.* Plenum Press, London.

ROLINE R.A. 1988. The effects of heavy metal pollution of the upper Arkansas river on the distribution of aquatic macroinvertebrates. *Hydrobiol.* **160**:3-8.

ROSENBERG D.M. & RESH V.H. 1993. *Freshwater biomonitoring and benthic macroinvertebrates.* Chapman & Hall, New York, U.S.A.

ROSENBERG D.M. & WIENS A.P. 1978. Effects of sediment addition on macrobenthic invertebrates in a northern Canadian river. *Wat. Res.* **12**: 753-763.

ROSICH & CULLEN. 1982. Nutrient Runoff. Pp. 103-119. In: *Water Quality management* (Ed. Hart B.T.). Water Studies Centre, Chisholm Institute of Technology & Australian Society for Limnology, Melbourne, Australia.

ROTHER H & LONDON L. 1998. Pesticide Health and Safety Policy Mechanisms in South Africa: The State of the Debate. Occupational & Environmental Health Research Unit, University of Cape Town. Working Paper No.1.

ROTHROCK J.A., BARTEN P.K. & INGMAN G.L. 1998. Land use and aquatic biointegrity in the Blackfoot River watershed, Montana. *Journal of the American Water Resources Association* **34** (3): 565-581.

ROUX D.J. 1990. *National water quality monitoring. The feasibility of implementing aquatic toxicity as part of surveillance monitoring in South Africa: a literature survey.* Hydrological Research Institute Report No. N 0000/00/REQ/2590, Department of Water Affairs & Forestry, Pretoria, South Africa. 51pp.

ROUX D.J. 1992. *Role and evaluation of biological monitoring in water quality assessment: a literature survey.* Hydrological Research Institute Report No. N 0000/00/REQ/1692, Department of Water Affairs & Forestry, Pretoria, South Africa. 76pp.

ROUX D., BADENHORST J.E., DU PREEZ H.H. & STEYN G.J. 1994. Nore on the occurrence of selected trace metals and organic compounds in water, sediment and biota of the Crocodile River, Eastern Transvaal, South Africa. *Water SA* **20:** 333- 430.

ROUX D.J., KEMPSTER P.L., KLEYNHANS C.J., VAN VLIET H.R. & DU PREEZ H.H. (1999). Integrating stressor and response monitoring into a resource-based water-quality assessment framework. *Environmental Management* **23** (1): 15-30.

ROUX, D.J., KEMPSTER P.L., TRUTER E. & VAN DER MERWE L. 1993. Effect of cadmium and copper on survival and reproduction of *Daphnia pulex*. *Water SA* 19: 269-274.

RUTHERFORD J.C., BLACKETT S., BLACKETT C., SAITO L., & DAVIES-COLLEY R.J. 1997. Predicting the effects of shade on water temperature in small streams. *N. Z. J. Mar. Freshwat. Res.* **31**: 707-721.

RUTTNER F. 1953. *Fundamentals of limnology.* University of Toronto Press, Toronto, Canada.

RYAN P.A. 1991. Environmental effects of sediment on New Zealand streams: a review. *N.Z. J. Mar. freshwater Res.* **25**: 207-221.

RYDER G.I. 1989. *Experimental studies on the effects of fine sediment on lotic invertebrates.* Unpublished PhD Thesis, Department of Zoology, University of Otago, Dunedin, New Zealand, 216p.

SABATER F., BUTTURINI A., MARTI E., MUNOZ I., ROMANI A., WRAY J. & SABATER S. 2000. Effects of riparian vegetation removal on nutrient retention in a Mediterranean stream. *J. N. Am. Benthol. Soc.* **19**(4): 609-620.

SAFRIEL O. & BRUTON M.N. 1984. *Aquaculture in South Africa: a cooperative research programme.* South African National Scientific Programmes Report No. 89., South Africa. 79pp.

SALLENAVE R.M. & BARTON D.R. 1990. The distribution of benthic invertebrates along a natural turbidity gradient in Lake Temiskaming, Ontario-Quebec. *Hydrobiol.* **206**: 225-234.

SANDERS H.O. 1970. Toxicities of some herbicides to six species of freshwater crustaceans. *J. wat. Poll. Contr. Fed.* **42**(8): 1544-1550.

SANDERSON R.D. & IMMELMAN E. 1990. *Research on the inhibition of bacterial oxidation of pyrite and the concomitant acid mine drainage: Part 3. Development and testing of slow release systems.* Water Research Commission Report No. 132/3/90. Pretoria, South Africa.

SANGFORS O. 1998. Are synergistic effects of acidification and eutrophication causing excessive algal growth in Scandanavian waters? *Ambio* **17**: 296.

SANTORE R.C., DI TORO D.M., PAQUIN P.R., ALLEN H.E. & MEYER J.S. 2001. Biotic ligand model of the acute toxicity of metals. 2. Application to acute copper toxicity in freshwater fish and *Daphnia*. *Environ Toxicol. Chem.* 20: 2397-2402.

SAVAGE N.C. & RABE F.W. 1973. *Northwest Sci.* **47**: 159.

SCARLETT G. & HARRIS J. 1990. Biological monitoring of a pulp and paper mill wastewater. *Environ. Monit. Assess.* **14**: 335-345.

SCARLETT G., HILL L. & ALLENDER B. 1985. Bioassay survey of the biological water quality of a pulp and paper mill effluent. *Appita* **40**: 104-107.

SCHERMAN P.A., PALMER C.G. & MULLER W.J. 2003. Use of indigenous riverine invertebrates in applied toxicology and water-resource quality management. *Water Research Commission Report* **955/1/03 ,** Water Research Commission, Pretoria.

SCHMIDT-NIELSEN K. 1990. (2nd Edition). *Animal Physiology*. Cambridge University Press.

SCHMITT C. J., RIBICK M.A., LUDKE J.L. & MAY T.W. 1983. National pesticide monitoring program: organochlorine residues in freshwater fish, 1976-1979. *US Fish and Wildlife Service, Resource Publication* **152**: 1-62.

SCHNEIDER J., MORIN A., & PICK F.R. 1995. The response of biota in experimental stream channels to a 24-hour exposure to the herbicide Velpar L®. *Environ. Toxicol. Chem.* **14:** 1607-1613.

SCHNOOR J.L. & STUMM W. 1985. Acidification of aquatic and terrestrial systems. Pp 311-338 *in* W. Stumm Ed.) *Chemical processes in lakes*. John Wiley & Sons, New York.

SCHOFIELD C.L. & TJORNAR J.R. 1980. Aluminum toxicity to fish in acidified waters. Pp 347-366 In: *Polluted rain*. (Eds. Toribara T.Y., Miller M.W. & Morrow P.E.), Plenum Press, New York.

SCHOFIELD N.J. & DAVIES P.E. 1996. Measuring the health of our rivers. *Water* May/June: 39-43.

SCHOFIELD N.J. & J.K. RUPRECHT. 1989. Regional analysis of stream salinization in southwestern Western Australia. *J. Hydrol.* **112**: 19-39.

SCHOLFIELD K., SEAGER J. & MERRIMAN R.P. 1990. The impact of intensive dairy farming activities on river quality: the Eastern Cleddau catchment study. *J. Inst. Wat. environ. Manag.* **4**: 176-186.

SCHOLTZ S., SEAMAN M.T. & PIETERSE A.J.H. 1988. Effects of turbidity on life history parameters of two species of *Daphnia*. *Freshwater Biol.* **20**: 177-184.

SCHUBAUER-BERIGAN M.K., MONSON P.D., WEST C.W. & ANKLEY G.T. 1995. Influence of pH on the toxicity of ammonia to *Chironomus tentans* and *Lumbriculus variegates*. *Env. Tox. Chem.* **14**(4): 713-717.

SCHULZ R. 1998. Macroinvertebrate dynamics in a stream receiving insecticide-contaminated runoff. *Verh. Internat. Verein. Limnol.* **26:** 1271-1276.

SCHULZ R. 2001. Comparison of spray drift- and runoff-related input of azinphos-methyl and endosulfan from fruit orchards into the Lourens River, South Africa. *Chemosphere* **45:** 543-551.

SCHULZ R. & DABROWSKI J.M. 2001. Combined effects of predatory fish and sublethal pesticide contamination on the behaviour and mortality of mayfly nymphs. *Environ. Toxicol. Chem.* **20:** 2537-2543.

SCHULZ R. & LIESS M. 1995. Chronic effects of low insecticide concentrations on freshwater caddisfly larvae. *Hydrobiol.* **299:** 103-113.

SCHULZ R. & PEALL S.K.C. 2001. Effectiveness of a constructed wetland for retention of nonpoint-source pesticide pollution in the Lourens River catchment, South Africa. *Environ. Sci. Technol.* **35:** 422-426.

SCHULZ R., PEALL S.K.C., DABROWSKI J.M. AND REINECKE A.J. 2001a. Current-use insecticides, phosphates and suspended solids in the Lourens River, Western Cape, during the first rainfall event of the wet season. *Water SA* **27**: 65-70.

SCHULZ R., PEALL S.K.C., DABROWSKI J.M. & REINECKE A.J. 2001b. Spray deposition of two insecticides into surface waters in the South African orchard area. *J. Environ. Qual.* **30:** 814-822.

SCHULZ R., PEALL S.K.C., HUGO C. & KRAUSE V. 2001c. Concentration, load and toxicity of spraydrift-borne azinphos-methyl at the inlet and outlet of a constructed wetland. *Ecol. Eng.* **18:** 239-245.

SCHULTZ R.C., COLLETTI J.P., ISENHART T.M., SIMPKINS W.W., MIZE C.W. & THOMPSON M.L. 1995. Design and placement of a multi-species riparian buffer strip system. *Agroforestry Systems* **29**: 201-226.

SCRIMGEOUR G.J. & WICKLUM D. 1996. Aquatic ecosystem health and integrity: problems and potential solution. *J. N. Am. Benthol. Soc.* **15** (2): 254-261.

SCRIVENER J.C. & BROWNLEE M.J. 1989. Effects of forest harvesting on spawning gravel and incubation survival of chum (*Onchorhynchus keta*) and coho salmon (*O. kisutch*) in Carnation Creek, British Columbia. *Canadian Journal of Fisheries and aquatic Sciences* **46:** 681-696.

SCULLION J. & EDWARDS R.W. 1980. The effect of coal industry pollutants on the macroinvertebrate fauna of a small river in the South Wales coalfield. *Freshwater Biol.* **10**: 141-162.

SEAGER J. & ABRAHAMS R.G. 1990. The impact of storm sewage discharges on the ecology of a small urban river. *Wat. Sci. Tech.* **22**: 163-171.

SEAMAN M.T., P.J. ASHTON & W.D. WILLIAMS. 1991. Inland salt waters of southern Africa. *Hydrobiol.* **210**: 75-91.

SERVIZI J.A., GORDON R.W. & MARTENS D.W. 1987. Acute toxicity of Garlon 4 and Roundup herbicides to salmon, *Daphnia*, and trout. *Bull. environ. Contam. Toxicol.* **39**: 15-22.

SHAFRON M., CROOME R. & ROLLS J. 1990. Water Quality. Pp. 147-165. In: *The Murray*. (Eds. Mackay N. & Eastburn D.). Murray-Darling Basin Commission, Canberra, Australia.

SHAW T.L. & BROWN V.M. 1974. The toxicity of some forms of copper to rainbow trout. *Wat. Res. Arch.* **8**: 377-382.

SHAZILI N.A. & PASCOE D. 1986. Variable sensitivity of rainbow trout (*Salmo gairdneri*) eggs and alevins to heavy metals. *Bull. environ. Contam. Toxicol.* **36**: 468-474.

SHEEDY B.R., LAZORCHAK J.M., GRUNWALD D.J., PICKERING Q.H., PILLI A., HALL D. & WEBB R. 1991. Effects of pollution on freshwater organisms. *J. wat. poll. Contr. Fed.* **63**: 619-696.

*SHEEHAN P.J. 1980. *The ecotoxicology of copper and zinc: studies on a stream macroinvertebrate community*. Ph.D. Dissertation, University of California, Davis, California, 262 pages.

SHEEHAN P.J. & WINNER R.W. 1984. Comparison of gradient studies in heavy-metal polluted streams. Pp. 255-272. In: *Effects of pollutants at the ecosystem level* (Eds. Sheehan P.J., Miller D.R. & Butler G.C.). Scientific commitee on problems of the environment. SCOPE 22, John Wiley & Son.

SHOESMITH E.A. & BROOK A.J. 1983. Monovalent-divalent cation ratios and the occurrence of phytoplankton, with special reference to desmids. *Freshwater Biol.* **13**: 151-155.

SILBERBAUER M.J. & KING J.M. 1991a. Geographical trends in the water chemistry of wetlands in the south-western cape Province, South Africa. *Sthn. Afr. J. aquat. Sci.* **17**: 82-88.

SILBERBAUER M.J. & KING J.M. 1991b. The distribution of wetlands in the south-western Cape Province, South Africa. *Sthn. Afr. J. aquat. Sci.* **17**: 65-81.

SIMPSON D.E. & HEMENS J. 1978. Nutrient budget for a residential stormwater catchment in Durban South Africa. *Prog. Wat. Tech.* **10**(5/6): 631-643.

SIMPSON J.C. &. NORRIS R.H 2000. Biological assessment of river quality: developments of AUSRIVAS models and outputs. In: *Assessing the biological quality of fresh waters: RIVPACS and other techniques* (Eds. Wright J.F., Sutcliffe D.W. & Furse M.T.), Freshwater Biological Association, United Kingdom.

SIMPSON K.W., BODE R.W. & COLQUHOUN J.R. 1985. The macroinvertebrate fauna of an acid-stressed headwater stream system in the Adirondack Mountains, New York. *Freshwater. Biol.* **15**: 671-681.

SINGH A.K. & GAUR J.P. 1989. Algal epilithon and water quality of a stream receiving oil refinery effluent. *Hydrobiol.* **184**: 193-199.

SIOLI H. 1955. Beiträge zur regionalen Limnologie der Amazonas-gebietes. III: Uber einige Gewässer des oberen Rio Negro-Gebietes. *Arch. f. Hydrobiol.* **50**: 1-32.

SKOROSZEWSKI R. 1989. Acid rain in South Africa. *African Wildlife* **43**: 82-83.

SLABBERT J.L. 1986. Improved bacterial growth test for rapid water toxicity screening. *Bull. environ. Contam. Toxicol.* **37**: 565-569

SLABBERT J.L. 1988. Microbial toxicity assays used for water quality evaluation in South Africa. *Toxicity Assessment: an international Quarterly*. **3**: 101-115.

SLABBERT J.L. & GRABOW W.O.K. 1986. A rapid water toxicity screening test based on oxygen uptake of *Pseudomonas putida*. *Toxicity Assessment: an international Quarterly*. **1**: 13-26.

SLABBERT J.L. & MAREE J.P. 1986. Evaluation of interactive effects of chemicals in water using *Tetrahymena pyriformis* screening test. *Water S.A.* **12**: 57-61.

SLABBERT J.L. & MORGAN W.S.G. 1982. A bioassay technique using *Tetrahymena pyriformis* for the rapid assessment of toxicants in water. *Wat. Res.* **16**: 517-523.

SLABBERT J.L., OOSTHUIZEN J., VENTER E.A., HILL E., DU PREEZ M. & PRETORIUS P.J. 1998. *Development of procedures to assess Whole Effluent Toxicity*. Water Research Commission Report No. 453/1/98. Water Research Commission, Pretoria, South Africa.

SLABBERT J.L., SMITH R. & MORGAN W.S.G. 1983. Application of *Tetrahymena pyriformis* bioassay system for the rapid detection of toxic substances in wastewaters. *Water S.A.* **9**: 81-87.

SLOANE-RICHEY J., PERKINS M.A. & MALUEG K.W. 1981. The effects of urbanisation and stormwater runoff on the food quality in two salmonid streams. *Verh. Internat. Verein. Limnol.* **21**: 812-818.

SMITH B.D., LYLE A.A. & MAITLAND P.S. 1983. The ecology of running waters near Aberfeldy, Scotland, in relation to a proposed barytes mine: an impact assessment. *Environ. Poll.* **32**: 269-306.

SMITH M.E. & KASTER J.L 1983. Effect of rural highway runoff on stream benthic macroinvertebrates. *Environ. Poll.* **32**: 157-170.

SMITH M.J., KAY W.R., EDWARD D.H.D., PAPAS P J., RICHARDSON K.J.,. SIMPSON J.C, PINDER A.M., CALE D.J., HOWITZ P.H.J., DAVIS J.A., YUNG F.H., NORRIS R.H. & HALSE S.A. 1999. AusRiVAS: using macroinvertebrates to assess ecological condition of rivers in Western Australia. *Freshwat. Biol.* **41** (2): 269-2.

SMITH S.V. & HOLLIBAUGH J.T. 1997. Annual cycle and interannual variability of ecosystem metabolism in a temperate climate embayment. *Ecol. Monog.* **67**: 509-533.

SMOCK L.A. & GAZZERA S.B. 1996. Effects of experimental episodic acidification on a southeastern USA blackwater stream. *J. Freshw. Ecol.* **11**: 81-90.

SNADDON C.D. & DAVIES B.R. 1998. A preliminary assessment of the effects of a small inter-basin transfer of water on discharge and invertebrate community structure. *Reg. Rivers Res. Manag.* **14**: 421-441.

SNADDON C.D. & DAVIES B.R. 1999. *An assessment of the ecological effects of inter-basin water transfer schemes (IBTs) in dryland environments.* Water Research Commission Report No. 665/1/00. Water Research Commission, Pretoria, South Africa.

SNADDON C.D., WISHART M. & DAVIES B.R. 1999. Some implications of inter-basin waster transfers for river functioning and water resources management in Southern Africa. *Aq. Ecosys. Health Manag.* **1**: 159-182.

SNADDON C.D., DAVIES B.R. & WISHART M. 2000. *A global overview of inter-basin waster transfer schemes, with an appraisal of their ecological, socio-economic and socio-political implications and recommendations for their management.* Water Research Commission Report No. TT 120/00. Water Research Commission, Pretoria, South Africa.

SOULSBY C., TURNBULL D., HIRST D., LANGAN S.J. & OWEN R. 1997. Reversibility of stream acidification in the Cairngorm region of Scotland. *J. Hydrol.* **195**: 291-311.

SOUTH AFRICA. 1956. Water Act: *Act 36 of 1956.*

SOUTH AFRICA. 1998. National Water Act: *Act 38 of 1998.*

SOUTHGATE B.A. 1951. Pollution of streams: some notes on recent research. *Inst. Civil Engineers Public Health Paper* No. 1

SPEHAR R.L., ANDERSON R.L. & FIANDT J.T. 1978. Toxicity and bioaccumulation of cadmium and lead in aquatic invertebrates. *Environ. Poll.* **15**:195-208.

SPEHAR R.L., CHRISTENSEN G.M., CURTIS C., LEMKE, NORBERG T.J. & PICKERING Q.H. 1982. Effects of pollution on freshwater fish. *J. wat. Poll. Contr. Fed.* **54**: 877-922.

SPEHAR R.L., FIANDT J.T., ANDERSON R.L. & DEFOE D.L. 1980. Comparative toxicity of arsenic compounds and their accumulation in invertebrates and fish. *Arch. environ. Contam. Toxicol.* **9**: 53-63.

SPRAGUE J.B. 1964. Lethal concentrations of copper and zinc for young Atlantic salmon. *J. Fish. Res. Bd. Can.* **21**: 17-26.

SPRAGUE J.B. 1971. Measurement of pollutant toxicity to fish. III. Sublethal effects and safe concentrations. *Wat. Res.* **5**: 245-266.

SPRAGUE J.B. & RAMSAY B.A. 1965. Lethal effects of mixed copper-zinc solutions for juvenile salmon. *J. Fish. Res. Bd. Can.* **22**: 425-432.

SRIDHAR M.K.C. 1986. Trace element composition of *Pistia stratiotes* L in a polluted lake in Nigeria. *Hydrobiol.* **131**: 273-276.

*STATE POLLUTION CONTROL COMMISSION. 1984. Sand and gravel extraction in the Upper Hawkesbury River. *ISBN* **0 7240 6000 6.**

STEINMAN A.D., MULHOLLAND P.J., PALUMBO A.V. & DEANGELIS D.L. 1992. *Ecol. App.* **2:** 341-355.

STENDELL R.C., GILMER D.S., COON N.A. & SWINEFORD D.M. 1988. Organochlorine and mercury residues in Swainson's and Ferrugineous Hawk eggs collected in North and South Dakota, 1974-1979. *Environ. Monit. Ass.* **10:** 37-41.

STEPHENSON D. & GREEN I.R.A. 1988. A simplistic mass balance of storm-water pollutants for two urban catchments. *Water S.A.* **14**: 93-98.

STEVEN J.D., DAVIES L.J., STANLEY E.K., ABBOTT R.A., INHAT M., BIDSTRUP L. & JAWORSKI J.F. 1976. *Effects of cadmium in the Canadian environment.* Nat. Res. Coun. Canada, NRCC No. 15017. 168pp.

STEWART N.E., SHUMWAY D.L. & DOUDOROFF P. 1967. Influence of oxygen concentration on growth of juvenile largemouth bass. *J. Fish. Res. Bd. Can.* **24**: 475-494.

STUMM W. & MORGAN J.J. 1970. *Aquatic Chemistry.* Wiley Interscience, New York.

SUTCLIFFE D.W. 1983. Acid precipitation and its effects on aquatic systems in the English Lake District (Cumbria). *Freshwater. Biol. Assoc.*, 51st Annual Report: 30-62.

SUZUKI K.T., KAWAHARA S., SUNAGA H., KOBAYASHI E. & SHIMOJO N. 1990. Discriminate uptake of metals by the liver and its relation to induction of metallothionein by cadmium, copper and zinc. *Comp. Biochem. Physio.* **95C**: 279-284.

SVENSSON B-G., BJORNHAM A., SHUTZ A., LETTEVALL U. & NILSSON A. 1987. Acidic deposition and human exposure to toxic metals. *Sci. tot. Environ.* **67**: 101-115.

SWIFT M.C. 1985. Effects of coal pile runoff on stream quality and macroinvertebrtae communities. *Wat. Res. Bull.* **21**: 449-457.

SZCZESNY B. 1974. The effect of sewage from the town of Krynica on the benthic invertebrates communities of the Kryniczanka stream. *Acta Hydrobiol.* **16**: 1-29.

SZCZESNY B. 1990. Benthic macroinvertebrates in acidified streams of the Swietokrzyski National Park (Central Poland). *Acta Hydrobiol.* **32**: 155-169.

TANKARD A.J., M.P.A. JACKSON, K.A. ERIKSSON, D.K. HOBDAY, D.R. HUNTER & W.E.L. MINTER. 1982. *Crustal evolution of South Africa.* Springer-Verlag.

TARZWELL C.M. 1970. Thermal requirements to protect aquatic life. *J. Wat. Poll. Contr. Fed.* **42**: 824-828.

TARZWELL C.M. & HENDERSON C. 1956. Toxicity of less common metals to fishes. *Indus. Wastes* **5**: 12.

TAYLOR B.R. & ROFF J.C. 1986. Longterm effects of highway construction on the ecology of a southern Ontario stream. *Environ. Poll.* (Series A) **40**: 317-344.

TECKLEMANN U. 1974. The effects of temperature on growth and metabolism of flowing water cold-stenotherms. *Arch. Hydrobiol.* **74**: 479-495, 516-522.

TELANG, S.A., BAKER B.L., HODGSON G.W., COSTERTON J.W., GEESEY G.G., MUTCH R.A. & WALLIS P. 1976. *Water quality and forest management: chemical and biological processes in a forest-stream ecosystem of the Marmot Creek drainage basin.* Inland Water Directorate, Environment Canada, Ontario and Environmental Planning and Research Services, Alberta, Canada.

TEMPORETTI P.F. & PEDROZO F.L. 2000. Phosphorus release rates from freshwater sediments affected by fish farming. *Aquaculture Research* **31:** 447-455.

THARME R.E. & KING J.M. 1998. *Development of the Building Block Methodology for instream flow assessments and supporting research on the effects of different magnitude flows on riverine ecosystems.* Water research Commission Report No. 576/1/98. Pretoria, South Africa. 452pp.

THRELKELD S.T. & SOBALLE D.M. 1988. Effects of mineral turbidity on freshwater planktonic communities: three exploratory tank experiments of factorial design. *Hydrobiol.* **159**: 223-236.

TILMAN D. 1999. Global environmental impacts of agricultural expansion: The need for sustainable and efficient practices. *Proc. Natl. Acad. Sci.* **96**: 5995-6000.

TRUTER E. 1990. *Toxicity assessment using Daphnia pulex as a test organism.* Hydrological Research Institute Report, Department of Water Affairs & Forestry, Pretoria, South Africa. 11pp.

TRUTER E. 1991a. *Pseudomonas putida growth inhibition test for water toxicity screening.* Hydrological Research Institute Report No. N0000/00/OEQ/3091, Department of Water Affairs & Forestry, Pretoria, South Africa. 9pp.

TRUTER E. 1991b. *Freshwater algal growth inhibition test with Selenastrum capricornutum.* Hydrological Research Institute Report No. N 0000/00/OEQ/3791, Department of Water Affairs & Forestry, Pretoria, South Africa. 9pp.

TRUTER E. 1991c. *Toxicity assessment using Poecilia reticulata (Guppy) as a test organism.* Hydrological Research Institute Report R198, Department of Water Affairs & Forestry, Pretoria, South Africa. 8pp.

TUCKER C.S., KINGSBURY S.K., POTE J.W. & WAX C.L. 1996. Effects of water management practices on discharge of nutrients and organic matter from channel catfish (*Ictalurus punctatus*) ponds. *Aquaculture* **147**: 57-69.

TULASI S.J. 1990. Ionic balance in the haemolymph of the freshwater crab *Barytelphusa guerini* (H Milne Edwards) exposed to sublethal concentrations of lead acetate and lead nitrate. *J. environ. Biol.* **11**: 163.

TULASI S.J., YASMEEN R. & RAMANA RAO J.V. 1985. Toxicity of organic and inorganic lead in fresh water crab *Barytelphusa guerini. J. Aquat. Biol.* **3**: 38-40.

TULASI S.J., YASMEEN R., REDDY C.P. & RAMANA RAO J.V. 1987. Lead uptake and lead loss in the fresh water field crab, *Barytelphusa guerini*, on exposure to organic and inorganic lead. *Bull. environ. Contam. Toxicol.* **39**: 63-68.

TURPENNY A.W.H. & WILLIAMS R. 1981. Factors affecting the recovery of fish populations in an industrial river. *Environ. Poll.* **26**: 39-58.

TYLER C., SUMPTER J. & JOBLING S. 1995. Environmental oestrogens and sexual development in fish. *Freshwat. Forum* **5:** 154-157.

TYSON P.D., KRUGER F.J., LOUW C.W. 1988. Atmospheric pollution and its implications in the Eastern Transvaal Highveld. *S.A. National Scientific Programmes Report No. 150.*, Council for Scientific & Industrial Research, Pretoria, 114pp.

ULRICH K.E., BURTON T.M. & UZARSKI D.G. 2000. The effects of whole-tree harvest on benthic insects in small New Hampshire streams. *Verh. Internat. Verein. Limnol.* **27**: 1079-1089.

USEPA 1986. *Quality criteria for water - 1986.* United States Environmental Protection Agency. Washington D.C.

VAN DER MERWE C.G., SCHOONBEE H.J. & PRETORIUS J. 1989. Observations on concentration of the heavy metals zinc, manganese, nickel, and iron in the water, in the sediments and in two aquatic macrophytes, *Typha capensis* (Rohrb.) N.E. Br and *Arundo donax* L., of a stream affected by goldmine and industrial effluent. *Water S.A.* **16**: 119-124.

VAN DER MERWE W. & GROBLER D.C. 1990. Water quality management in the RSA: preparing for the future. *Water S.A.* **16**: 49-54.

VAN DER PUTTE I., LUBBERS J. & KOLAR Z. 1981. Effect of pH on uptake, tissue distribution and retention of hexavalent chromium in rainbow trout *Salmo gairdneri*. *Aquat. Toxicol.* **1**: 3-18.

VICTOR R. & OGBEIBU A.E. 1986. Recolonization of macrobenthic invertebrates in a Nigerian stream after pesticide treatment and associated disruption. *Environ. Poll.* (Series A) **41**: 125-137.

VINYARD G.L. 1996. A chemical and biological assessment of water quality impacts from acid mine drainage in a first order mountain stream, and a comparison of two bioassay techniques. *Environ. Tech.* **17**: 273 - 281.

VOYER R.A. & MCGOVERN D.G. 1991. Influence of constant and fluctuating salinity on responses of *Mysidopsis bahia* exposed to cadmium in a life-cycle test. *Aquat. Toxicol.* **19**: 215-230.

VOYER R.A. & MODICA G. 1990. Influence of salinity and temperature on acute toxicity of cadmium to *Mysidopsis bahia* Molenock. *Arch. environ. Contam. Toxicol.* **19**: 124-131.

VUORI K. & JOENSUU I. 1996. Impact of forest drainage on the macroinvertebrates of a small boreal headwater stream: do buffer zones protect lotic biodiversity? *Biol. Cons.* **77**: 87-95.

VUORI K.-M. 1994. Rapid behavioural and morphological responses of hydropsychid larvae (Trichoptera, Hydropsychidae) to sublethal cadmium exposure. *Environ. Poll.* **84:** 291-299.

Vuori K.-M. 1995. Species- and population-specific responses of translocated hydropsychid larvae (Trichoptera, Hydropsychidiae) to runoff from acid sulphate soils in the River Kyrönjoki, western Finland. *Freshwat. Biol.* **33**: 305-318.

VYMAZAL J. 1987. Toxicity and accumulation of cadmium with respect to algae and cyanobacteria: a review. *Toxicity Assessment: An International Quarterly Review* **2**: 387-415.

WALDEN C.C. 1976. Review paper: the toxicity of pulp and paper mill effluents and corresponding measurement procedures. *Wat. Res.* **10**: 639-664.

WALLACE J.B. & GURTZ M.E. 1986. Response of *Baetis* mayflies (Ephemeroptera) to catchment logging. *Am. Midl. Nat.* **115**: 25-41.

WALLACE J.B., GRUBAUGH J.W. & WHILES M.R. 1996. Biotic Indices and stream ecosystem processes: Results from an experimental study. *Ecol. App.* **6** (1): 140-151.

WALLACE J.B., HURYN A.D. & LUGTHART G.T. 1991. olonization of a headwater stream during three years of seasonal insecticidal applications. *Hydrobiol.* **211**: 65-76.

WALLACE J.B., VOGEL D.S. & CUFFNEY T.F. 1986. Recovery of a headwater stream from an insecticide-induced community disturbance. *J. N. Am. Benthol. Soc.* **5**: 115-126.

*WALLEN J., WIGHTMAN A.S. & PENNING W. 1984. Environmental effects of cadmium in fresh water: A bibliography. *Freshwater Biological Association Occasional Publication* **No 23**.

WALMSLEY R.D. & BUTTY M. 1980. *Guidelines for the control of eutrophication in South Africa.* Water Research Commission - National Institute for Water Research Collaborative Report, Pretoria, South Africa. ISBN 0-7988-1735-6.

WALSH C.J. 2000. Urban impacts on the ecology of receiving waters: a framework for assessment, conservation and restoration. *Hydrobiol.* **131**: 107-111.

WALSCH C.J. & BREEN P.F. 2001. A biological approach to assessing the potential success of habitat restoration in urban streams. *Ver. Int. Ver. Limnol.* **27:** 3654-3658.

WALSCH C.J., SHARPE A.K., BREEN P.F. & SONNEMAN J.A. 2001. Effects of urbanisation on streams of the Melbourne region, Victoria, Australia. I. Benthic macroinvertebrate communities. *Freshwater Biology* **46** (4): 535-551.

WARD J.V. 1985. Thermal characteristics of running waters. *Hydrobiol.* **125**: 31-46.

WARD J.V. 1998. Riverine landscapes: biodiversity patterns, disturbance regimes , and aquatic conservation. *Biological Conservation* **83** (3): 269-278.

WARD J.V. & STANFORD J.A. 1982. Thermal responses in the evolutionary ecology of aquatic insects. *Ann. Rev. Entomol.* **27**: 97-117.

WARD J.V., CANTON S.P. & GRAY L.J. 1978. The stream environment and macroinvertebrate communities: contrasting effects of mining in Colorado and the eastern United States. Pp. 176-187. In: *Energy and environmental stress in aquatic systems.* (Eds. Thorp J.H. & Gibbons J.W.). DOE Symposium Series (CONF-771114), Springfield, Va.

WARNICK S.L. & BELL H.L. 1969. The acute toxicity of some heavy metals to different species of aquatic insects. *J. wat. Poll. Contr. Fed.* **41**: 280-284.

WARREN C.E. 1971. *Biology and water pollution control.* W.B. Saunders Co., Philadelphia.

WATLING R.J. & EMMERSON W.D. 1981. A preliminary pollution survey of the Papenkuils River, Port Elizabeth. *Water S.A.* **7**: 211-215.

WATSON S.J. & MALY E.J. 1987. Thiocyanate toxicity to *Daphnia magna*: modified by pH and temperature. *Aquat. Toxicol.* **10**: 1-8.

WEATHERLEY N.S. *et al.* 1990. The survival of early life stages of brown trout (*Salmo trutta* L.) in relation to aluminium speciation in upland Welsh streams. *Aquat. Toxicol.* **17**: 213.

WEAVER J.M.C 1993. A Preliminary Survey of Pesticide Levels in Groundwater from a selected area of intensive agriculture in the Western Cape. Water Research Commission Report 268/1/93. Water Research Commission, Pretoria, South Africa.

*WEBER C.I. 1981. Evaluation of the effects of effluents on aquatic life in receiving waters - an overview. Pp. 3-13. In: *Ecological assessments of effluent impacts on communities of indigenous aquatic organisms* (Eds. Bates J.M. & Weber C.I.). American Society for Testing and Materials STP 730.

*WEBER C.I. & MCFARLAND B.H. 1981. Effects of copper on periphyton of a small calcareous stream. Pp. 101-131. In: *Ecological assessments of effluent impacts on communities of indigenous aquatic organisms* (Eds. Bates J.M. & Weber C.I.). American Society for Testing and Materials STP 730.A

WEBSTER I.A., FORD P.W. & HANCOCK G. 2001. Phosphorus dynamics in Australian lowland rivers. *Mar. Freshwat Res.* 52: 127-137.

WEEKS C.R. 1982. Pollution in urban runoff. Pp. 121-139. In: *Water quality management: monitoring programs and diffuse runoff.* (Ed. Hart B.T.). Water Studies Centre, Chisholm Institute of Technology & Australian Society for Limnology, Melbourne, Australia.

WEITZEL R.L. & BATES J.M. 1981. Assessment of effluents through evaluation of periphyton diatom community structure. Pp. 142-165. In: *Ecological assessments of effluent impacts on communities of indigenous aquatic organisms* (Eds. Bates J.M. & Weber C.I.). American Society for Testing and Materials STP 730.

WELLBORN G.A. & ROBINSON J.V. 1996. Effects of a thermal effluent on macroinvertebrates in a central Texas reservoir. *American Midland Naturalist* **136**(1): 110-120.

WELLNITZ T.A. & SHELDON S.P. 1995. The effects of iron and manganese on diatom colonization in a Vermont stream. *Freshwat. Biol.* **34**: 465-470.

WELLS J. 1992. *A pre-impoundment study of the biological diversity of the benthic macro-invertebrate fauna of the Sabie-Sand River System.* Unpublished M.Sc. Thesis, University of Cape Town, South Africa. 207pp.

WELLS J.D. 1987. State of the art: The closure and rehabilitation of mining waste impoundments. Pp. 337-343. In: *Proceedings of the International Conference on Mining and Industrial Waste Manangement.* Johannesburg, South Africa.

WENDELAAR B., BONGA S.E. & DEDEREN L.H.T. 1986. Effects of acidified water on fish. *Endeavour* **10**: 198-202.

WEST G., BLYTH J.D. BALKAU F. & SLATER S. 1984. Chemical and physical features of construction sediments in the Mitta Mitta River, Victoria. *Occ. Pap. Mus. Vict.* **1**: 129-138.

WHITE P.S. & PICKETT S.T.A. 1985. Natural disturbance and patch dynamics: an introduction. In: *The ecology of natural disturbance and patch dynamics* (Eds. Pickett S.T.A & White P.S.), Academic Press, New York, U.S.A.

WHITEHURST I.T. & LINDSEY B.I. 1990. The impact of organic enrichment on the benthic macroinvertebrate communities of a lowland river. *Wat. Res.* **24**: 625-630.

WHITING E.R., MATHIEU S. & PARKER D.W. 1994. Effects of drainage from a molybdenum mine and mill on stream macroinvertebrate communities. *J. Freshwat. Ecol.* **9**: 299-311.

WHITLEY L.S. 1968. The resistance of tubificid worms to three common pollutants. *Hydrobiol.* **32**: 193-205.

WIGINGTON P.J., DAVIES T.D., TRANTER M. & ESHLEMAN K.N. 1992. Comparison of episodic acidification in Canada, Europe and the United States. *Environ. Poll.* **78**: 29-35.

WILBUR W.G. & HUNTER J.V. 1979. Distribution of metals in street sweepings, stormwater solids, and urban aquatic sediments. *J. wat. Poll. Contr. Fed.* **51**: 2810-2822.

WILCOCK R.J., COSTLEY K.J., COWLES R.J., WILSON B. & SOUTHGATE P. 1991. Stream run-off losses and soil and grass residues of triclopyr applied to hillside gorse. *N. Z. J. Agric. Res.* **34**: 351-357.

WILHM T.P. & DORRIS T.C. 1968. Biological parameters for water quality criteria. *Bio. Sci.* **18**: 477-481.

WILLIAMS K.A., GREEN D.W.J. & PASCOE D. 1986. Studies on the acute toxicity of pollutants to freshwater macroinvertebrates. *Arch. Hydrobiol.* **106**: 61-70.

WILLIAMS K.A., GREEN D.W.J., PASCOE D. & GOWER D.E. 1987. Effect of cadmium on oviposition and egg viability in *Chironomus riparius* (Diptera: Chironomidae). *Bull. environ. Contam. Toxicol.* **38**: 86-90.

WILLIAMS M.D. & WILLIAMS W.D. 1991. Salinity tolerances of four fish species from the Murray-Darling River system. *Hydrobiol.* **210**: 145-160.

WILLIAMS W.D. 1981. Inland salt lakes: an introduction. *Hydrobiol.* **81/82**: 1-14.

WILLIAMS W.D. 1987. Salinization of rivers and streams: an important environmental hazard. *Ambio* **16**: 180-185.

WILLIAMS W.D., TAAFFE R.G. & BOULTON A.J. 1991. Longitudinal distribution of macroinvertebrates in two rivers subject to salinization. *Hydrobiol.* **210**: 151-160.

WILSON A.L. 1976. *Concentrations of trace metals in river waters: a review.* Technical Report 16, Treatment Division, Water Research Centre, United Kingdom. 60pp.

WINNER R.W., BOESEL M.W. & FARRELL M.P. 1980. Insect community structure as an index of heavy-metal pollution in lotic ecosystems. *Can. J. Fish. Aquat. Sci.* **37**: 647-655.

WINNER R.W., VAN DYKE J.S., CARIS N. & FARREL M.P. 1975. Response of the macroinvertebrate fauna to a copper gradient in an experimentally-polluted stream. *Ver. Intern. Verein Limnol.* **19**: 2121-2127.

WISHART M.J., DAVIES B.R., STEWART B.A. & HUGHES J.M. 2003. *Examining catchments as functional units for the conservation of riverine biota and maintenance of biodiversity.* Final report to the Water Research Commission, Pretoria, South Africa.

WISHART, M.J. & J.A. DAY. 2002. Endemism in the freshwater fauna of the south-western Cape, South Africa. *Verhandlungen, internationale Vereinigung für theoretische und angewandte Limnologie.*

WONG C.K. & WONG P.K. 1990. Life table evaluation of the effects of cadmium exposure on the freshwater cladoceran, *Moina macrocopa*. *Bull. environ. Contam. Toxicol.* **44**: 135-141.

WOOD P.J. & ARMITAGE P.D. 1997. Biological effects of fine sediment in the lotic environment. *Environ. Manag.* **21**(2): 203-217.

WOOD R.B. & TALLING J.F. 1988. Chemistry and algal relationships in a saline series of Ethiopian inland waters. *Hydrobiol.* **158**: 29-67

WOOD C.M. & ROGANO M.S. 1986. Physiological responses to acid stress in crayfish (*Orconectes*): hemolymph ions, acid-base status, and exchanges with the environment. *Can. J. Fish. Aquat. Sci.* **43**: 1017-1026.

WOOD H. 2002. Tackling the legacy of mining. *Water 21* February. 41-42.

WOODIWISS F.S. 1964. The biological system of stream classification used by the Trent River Board. *Chem. Ind.* **11**: 443-447.

WOODWELL G.M., WURSTER Jr.C.F. & ISAACSON P.A. 1967. DDT residues in an east coast estuary: a case of biological concentration of a persistent pesticide. *Science* **156**: 821-.

WRC (WATER RESEARCH COMMISSION)) 1990. *The chemical removal of sulphates*. Water Research Commission Report No. 203/1/90, Pretoria, South Africa.

WRENN W.B., ARMITAGE B.J., RODGERS E.B. & FORSYTHE T.D. 1979. Effects of temperature on bluegill and walleye, and periphyton, macroinvertebrates, and zooplankton communities in experimental ecosystems. *Brown's Ferry Biothermal Research Series II*. Duluth, MNB, Environmental Protection Agency Ecological Research EPA-600/3-79-092.

WRIGHT I.A., CHESSMAN B.C., FAIRWEATHER P.G. & BENSON L.J. 1995. Measuring impacts of sewage effluent on the macroinvertebrate community of an upland stream: the effect of different levels of taxonomic resolution and quantification. *Aust. J. Ecol.* **20**: 142-149.

WRIGHT J.F. 1995. Development and use of a system for predicting the macroinvertebrate fauna in flowing waters. *Aust. J. Ecol.* **20**: 181-197.

WRIGHT J.F., FURSE M.T., ARMITAGE P.D. 1993. RIVPACS- A technique for evaluating the biological quality of rivers in the U.K. *Freshwat. Biol.* **3** (4): 15-25.

WU Y.F. 1931. A contribution to the biology of *Simulium*. *Pap. Mich. Acad. Sci.* **13**: 543-599.

WUHRMANN K. 1976. Chemical impacts on inland aquatic ecosystems. *Pure & Appl. Chem.* **45**: 193-198.

YOUNGER P.L. 2001. Mine water pollution in Scotland: nature, extent and preventative strategies. *The Science of the Total Environment*. **265**: 309-326.

ZISCHE J.A., ARTHUR J.W., NORDLIE K.J., HERMANUTZ R.O., STANDEN D.A. & HENRY T.P. 1983. Acidification effects on macroinvertebrates and fathead minnows (*Pimephales promelas*) in outdoor experimental channels. *Wat. Res.* **17**: 47-63.

ZWEIG L.D. & RABENI C.F. 2001. Biomonitoring for deposited sediment using benthic invertebrates: a test of 4 Missouri streams. *J. N. Am. Benthol. Soc.* **20**(4): 643-657.

ZYGMUNTOWA J. 1988. The effect of nitrogen fertilizer of bankside soil on the chemical characteristics of the River Nida (Southern Poland) with special regard to dissolved organic matter. *Acta Hydrobiol.* **30**: 167-178.

Appendix A An alphabetic list of the scientific names of all organisms referred to in the text with details of their taxonomic classification and common name

SCIENTIFIC NAME	PHYLUM	CLASS	ORDER / FAMILY*	COMMON NAME
Acroneuria lycorias	Arthropoda	Insecta	Plecoptera	stonefly
Afrochiltonia capensis	Arthropoda	Crustacea	Amphipoda	scud/ sand hopper
Alburnus alburnus	Chordata	Osteichthyes		bleak (fish)
Anabaena flos-aquae				blue-green alga
Anabaena variabilis				blue-green alga
Anabaena				blue-green alga
Aplexa hypnorum	Mollusca	Gastropoda	Physidae*	snail
Artemia salina	Arthropoda	Crustacea		brine shrimp
Asellus aquaticus	Arthropoda	Crustacea	Isopoda	sow-bug/ hog-louse
Asellus communis	Arthropoda	Crustacea	Isopoda	sow-bug/ hog-louse
Asellus meridianus	Arthropoda	Crustacea	Isopoda	sow-bug/ hog-louse
Aulosira fertilissima				alga
Australorbis glabratus	Mollusca	Gastropoda	Planorbidae*	snail
Baetis bellus	Arthropoda	Insecta	Ephemeroptera	mayfly
Baetis latus	Arthropoda	Insecta	Ephemeroptera	mayfly
Baetis rhodani	Arthropoda	Insecta	Ephemeroptera	mayfly
Baetis thermicus	Arthropoda	Insecta	Ephemeroptera	mayfly
Baetisca laurentina	Arthropoda	Insecta	Ephemeroptera	mayfly
Barytelphusa guerini	Arthropoda	Crustacea	Brachyura	crab
Bellamya dissimilis	Mollusca	Gastropoda	Viviparidae*	snail
Boleophthalmus dussumieri	Chordata	Osteichthyes		mud skipper (fish)
Brachionus rubens	Rotifera			rotifer
Brachycentrus americanus	Arthropoda	Insecta	Trichoptera	caddisfly
Caenis	Arthropoda	Insecta	Ephemeroptera	mayfly
Carassius auratus	Chordata	Osteichthyes	Cypriniformes	goldfish
Catostomus commersoni	Chordata	Osteichthyes		white sucker (fish)
Ceriodaphnia dubia	Arthropoda	Crustacea	Cladocera	water flea
Ceriodaphnia pulchella	Arthropoda	Crustacea	Cladocera	water flea
Channa punctatus	Chordata	Osteichthyes		snakehead (fish)
Chaoborus	Arthropoda	Insecta		ghost midge/ phantom larva
Chasmagnathus granulata	Athropoda	Crustacea	Brachyura	crab
Chironomus plumosus	Arthropoda	Insecta	Diptera	midge
Chironomus riparius	Arthropoda	Insecta	Diptera	midge
Chironomus tentans	Arthropoda	Insecta	Diptera	midge
Chironomus	Arthropoda	Insecta	Diptera	midge
Chlorella pyrenoidosa				alga
Chlorella saccharophila				alga
Cladophora				alga
Clarius batrachus	Chordata	Osteichthyes	Siluriformes	catfish
Clarius gariepinus	Chordata	Osteichthyes	Siluriformes	sharptooth catfish
Cloeon	Arthropoda	Insecta	Ephemeroptera	mayfly

SCIENTIFIC NAME	PHYLUM	CLASS	ORDER / FAMILY*	COMMON NAME
Coretus corneus	Mollusca	Gastropoda		snail
Coretus fatigous	Mollusca	Gastropoda		snail
Cricotopus	Arthropoda	Insecta	Diptera	midge
Culex fatigous	Chordata	Insecta	Diptera	mosquito
Cyclatella meneghiniana				diatom
Cyclops	Arthropoda	Crustacea	Copepoda	minute "shrimp"
Cyprinus carpio	Chordata	Osteichthyes	Cypriniformes	common carp (fish)
Daphnia carinata	Arthropoda	Crustacea	Cladocera	water flea
Daphnia magna	Arthropoda	Crustacea	Cladocera	water flea
Daphnia pulex	Arthropoda	Crustacea	Cladocera	water flea
Ecdyonurus dispar	Arthropoda	Insecta	Ephemeroptera	mayfly
Ephemera simulans	Arthropoda	Insecta	Ephemeroptera	mayfly
Ephemerella grandis	Arthropoda	Insecta	Ephemeroptera	mayfly
Ephemerella ignita	Arthropoda	Insecta	Ephemeroptera	mayfly
Ephemerella subvaria	Arthropoda	Insecta	Ephemeroptera	mayfly
Ephemerella	Arthropoda	Insecta	Ephemeroptera	mayfly
Eristalis	Arthropoda	Insecta	Diptera	rat-tailed maggot
Erpobdella octoculata	Annelida	Hirudinae		leech
Gambusia affinis	Chordata	Osteichthyes	Cyprinodontiformes	mosquito fish
Gammarus fasciatus	Arthropoda	Crustacea	Amphipoda	scud/ sand hopper
Gammarus lacustris	Arthropoda	Crustacea	Amphipoda	scud/ sand hopper
Gammarus pseudolimnaeus	Arthropoda	Crustacea	Amphipoda	scud/ sand hopper
Gammarus pulex	Arthropoda	Crustacea	Amphipoda	scud/ sand hopper
Gasterosteus aculaeatus	Chordata	Osteichthyes		threespine stickleback (fish)
Heteropneustes fossilis	Chordata	Osteichthyes	Siluriformes	indian catfish
Hexagenia limbata	Arthropoda	Insecta	Ephemeroptera	mayfly
Hyallela azteca	Arthropoda	Crustacea	Amphipoda	scud/ sand hopper
Hydridella glenelgenis	Mollusca	Bivalvia		river mussel
Hydropsyche betteni	Arthropoda	Insecta	Trichoptera	caddisfly
Hydropsyche	Arthropoda	Insecta	Trichoptera	caddisfly
Hydrozetes	Arthropoda	Arachnida	Hydracarina	water mite
Ictalurus punctatus	Chordata	Osteichthyes	Siluriformes	channel catfish
Ischnura verticalis	Arthropoda	Insecta	Odonata	dragonfly
Jordanella floridae	Chordata	Osteichthyes		flagfish
Juga plicifera	Mollusca	Gastropoda		snail
Lebistes reticulatus	Chordata	Osteichthyes		guppy
Lemna gibba				duckweed
Lepomis cyanellus	Chordata	Osteichthyes	Perciformes	green sunfish
Lepomis macrochirus	Chordata	Osteichthyes	Perciformes	bluegill sunfish
Leptophlebia nebulosa	Arthropoda	Insecta	Ephemeroptera	mayfly
Leuctra inermis	Arthropoda	Insecta	Plecoptera	stonefly
Limnodrilus hoffmeisteri	Annelida	Oligochaeta	Tubificidae	aquatic earthworm
Lucioperca	Chordata	Osteichthyes	Perciformes	perch
Lymnaea stagnalis	Mollusca	Gastropoda	Lymnaeidae*	snail

SCIENTIFIC NAME	PHYLUM	CLASS	ORDER / FAMILY*	COMMON NAME
Macrobrachium rosenbergii	Arthropoda	Crustacea	Decapoda	freshwater prawn
Macrobrachium lamarrei	Arthropoda	Crustacea	Decapoda	freshwater prawn
Macromia	Arthropoda	Insecta	Odonata	dragonfly
Microcyctis aeruginosa				blue-green alga
Micropterus dolomieui	Chordata	Osteichthyes	Perciformes	smallmouth bass
Micropterus salmoides	Chordata	Osteichthyes	Perciformes	largemouth bass
Moina macrocopa	Arthropoda	Crustacea	Cladocera	water flea
Monoraphidium circinale				green alga
Morone saxatilis	Chordata	Osteichthyes	Perciformes	striped bass
Mysiodopsis batia	Arthropoda	Crustacea	Mysidacea	opossum shrimp
Nais	Annelida	Oligochaeta	Naididae	aquatic earthworm
Navicula incerta				alga
Nereis virens	Annelida	Polychaeta		bristle worm
Nitzschia closterium				alga
Notroptis spilopterus	Chordata	Osteichthyes		spotfin shiner (fish)
Oncorhynchus gorbuscha	Chordata	Osteichthyes	Salmonidae	pink salmon
Oncorhynchus keta	Chordata	Osteichthyes	Salmonidae	chum salmon
Oncorhynchus kisutch	Chordata	Osteichthyes	Salmonidae	coho salmon
Oncorhynchus mykiss	Chordata	Osteichthyes	Salmonidae	steelhead/ rainbow trout
Oncorhynchus nerka	Chordata	Osteichthyes	Salmonidae	sockeye salmon
Oncorhynchus tshawytscha	Chordata	Osteichthyes	Salmonidae	chinook salmon
Orconectes immunis	Arthropoda	Crustacea	Brachyura	freshwater crayfish
Oxyethira velocipes	Arthropoda	Insecta	Trichoptera	caddisfly
Oziotelphusa senex senex	Arthropoda	Crustacea	Brachyura	crab
Perca flavescens	Chordata	Osteichthyes	Perciformes	yellow perch
Perca fluvialus	Chordata	Osteichthyes	Perciformes	perch
Physa fontinalis	Mollusca	Gastropoda	Physidae*	snail
Physa gyrina	Mollusca	Gastropoda	Physidae*	snail
Physa heterostropha	Mollusca	Gastropoda	Physidae*	snail
Physa integra	Mollusca	Gastropoda	Physidae*	snail
Physa	Mollusca	Gastropoda	Physidae*	snail
Pimephales promelas	Chordata	Osteichthyes		fathead minnow
Planorbidae	Mollusca	Gastropoda	Planorbidae*	snail
Poecilia reticulata	Chordata	Osteichthyes	Atheriniformes	guppy
Polycelis tenuis	Platyhelminthes	Turbellaria	Tricladida	flatworm
Polypedilum anale	Arthropoda	Insecta	Diptera	midge
Pontogammarus maeoticus	Arthropoda	Crustacea	Amphipoda	scud/sand hopper
Pristina	Annelida	Oligochaeta	Naididae	aquatic earthworm
Pseudomonas putida				bacterium
Psychoda	Arthropoda	Insecta	Diptera	sewage fly
Pteronarcella badia	Arthropoda	Insecta	Plecoptera	stonefly
Pteronarcys californica	Arthropoda	Insecta	Plecoptera	stonefly
Pteronarcys dorsata	Arthropoda	Insecta	Plecoptera	stonefly
Salmo clarkii	Chordata	Osteichthyes	Salmonidae	cutthroat salmon

SCIENTIFIC NAME	PHYLUM	CLASS	ORDER / FAMILY*	COMMON NAME
Salmo fario	Chordata	Osteichthyes	Salmonidae	
Salmo gairdneri	Chordata	Osteichthyes	Salmonidae	rainbow trout
Salmo sala	Chordata	Osteichthyes	Salmonidae	atlantic salmon
Salmo trutta	Chordata	Osteichthyes	Salmonidae	brown trout
Salvelinus fontinalis	Chordata	Osteichthyes	Salmonidae	brook trout
Selenastrum capricornutum				alga
Simocephalus vetulus	Arthropoda	Crustacea	Cladocera	water flea
Simocephalus serrulatus	Arthropoda	Crustacea	Cladocera	water flea
Stizostedion vitreum	Chordata	Osteichthyes		walleye (fish)
Tanytarsus dissimilis	Arthropoda	Insecta	Diptera	midge
Thiara torulosa	Mollusca	Gastropoda	Thiaridae*	snail
Tilapia aurea	Chordata	Osteichthyes	Perciformes	blue tilapia
Tilapia milticus	Chordata	Osteichthyes	Perciformes	bream
Tilapia mossambicus	Chordata	Osteichthyes	Perciformes	tilapia
Tilapia sparrmanii	Chordata	Osteichthyes	Perciformes	banded tilapia
Tilapia spilurus	Chordata	Osteichthyes	Perciformes	bream
Tubifex tubifex	Annelida	Oligochaeta	Tubificidae	aquatic earthworm

GLOSSARY

abiotic	not related to living things
acclimation	the process whereby an organism becomes accustomed to artificially imposed conditions
acetylcholine	an enzyme involved in transmission of nervous impulses, and the target of organochlorine insecticides
acid precipitation	rain, mist, snow and dust acidified by pollution with oxides of nitrogen and/or sulphur
acid rain	see *acid precipitation*
acidification	here considered as an increase in the acidity of water and soil as a result of human activities
acute effect	in toxicology, the effect of a toxin during a short period of exposure
adsorption	the physical adherence to the surface of a molecule or particle
aeolian	of the wind
aerobic	using oxygen
agrochemicals	chemicals, such as pesticides and fertilisers, used in agriculture
algae	non-vascular plant-like organisms (seaweeds, pond scums, many kinds of phytoplankton, etc.)
alkalinity	the sum of the anions of weak acids, plus hydroxyl, carbonate and bicarbonate ions, in a sample of water
allochthonous	generated outside the system in question
ambient	surrounding (e.g. ambient temperature is temperature of the surrounding air or water)
Amphipoda	a group of small shrimp-like crustaceans (sand fleas, beach hoppers, scuds)
anaerobic	not using oxygen
anoxic	without oxygen
animal	any heterotrophic, motile organism (includes worms, sponges, birds, reptiles, humans, etc.)
anions	negatively charged ions
Annelida	the annulated worms, including earthworms, leeches and bristle worms
antagonist	one acting against another (in toxicology, a substance that reduces the toxic effect of another)
antecedent	preceding
antibiotic	an anti-bacterial agent
anthropogenic	caused by human activity
aquifer	a subterranean accumulation of water
Arachnida	the eight-legged arthropods: spiders, mites, ticks, scorpions, etc.
Arthropoda	the jointed-legged animals: the crustaceans, insects, arachnids, etc.
ATP	adenosine triphosphate, an energy-rich organic molecule
autotrophic	'self-feeding': using the Sun's energy or chemical energy to fix carbon (contrast with *heterotrophic*)
bacteria	minute prokaryotic organisms with no nucleus; some, but not all, bacteria are pathogenic (q.v.)
basalt	a rock type of volcanic origin
benthic	living on the bottom
bilharzia	schistosomiasis: a human parasitic disease caused by a fluke whose intermediate host is a freshwater snail
bioaccumulation	the accumulation of substances within a living organism
bioassay	an assay using living organisms

bioassessment	assessment, usually of environmental condition, using living organisms
bioavailability	the extent to which a particular constituent is available to living organisms
biocides	substances that kill living organisms (usually substances created by humans for this purpose)
bioconcentration	accumulation of a chemical substance in a living organism by direct contact with the surrounding medium
biodegradable	able to be broken down by living organisms, often bacteria
biodiversity	the diversity of living things at the level of genetics, of whole organisms, and of ecosystems
biotic assemblage	the species found in an area
biological indicator	living things that indicate the quality of environmental conditions
biological community	all the interacting members of the biota of a given area
biological oxygen demand (= BOD)	the amount of oxygen consumed by the biota in a sample of water (usually measured over 5 days at $20^{\circ}C$)
biomagnification	the phenomenon resulting from accumulation of chemical substances in living organisms through the food chain
biomass	the mass (weight) of biological material
biomonitoring	using living organisms to monitor aspects of their environment
biota	the totality of the living organisms of a region or a system
biotic	relating to living things
biotope	an homogeneous environment that satisfies the habitat *requirements of a biotic community*
Bivalvia	molluscs with two shells: clams, mussels, etc.
blackflies	blood-sucking midges with filter-feeding aquatic larvae that in parts of Africa are vectors of river blindness
blackwater systems	aquatic systems in which the water is stained with humic, fulvic and/or other polyphenolic acids
BOD	(see *biological oxygen demand*)
Brachyura	crabs
buffering capacity	the resistance to change in pH as a result of the presence in water of a weak acid and its salts
caddisfly	a member of the insect order Trichoptera, the larvae of many of which are aquatic
canopy	the leafy parts of trees meeting overhead
carcinogenic	causing cancer
carnivore	a meat-eater
catchment	the entire land area from which a river or reservoir is fed (= drainage basin, = watershed)
cations	positively-charged ions
chemical oxygen demand (COD)	the amount of oxygen consumed by the abiotic fraction of a water sample
Chironomidae	a family of non-biting midges, the larvae of which are very common inhabitants of fresh waters
chronic effect	in toxicology, the effect of a toxin over a long period of exposure
Ciliata	protozoans (*q.v.*), such as *Paramecium*, that moves by means of long hair-like structures called cilia
Cladocera	the water fleas
Cladophora	a genus of filamentous alga
COD	(see *chemical oxygen demand*)
Coleoptera	beetles
coliform (bacteria)	bacteria related to *Escherischia coli*, which is found in human faeces

collector	an invertebrate animal that collects small particles of food (includes filter feeders and deposit feeders)
community	biologically, the interacting species living together in a particular area
conductivity	the ability of water to conduct an electrical current; since this depends on the number of ions in solution, also a measure of the total quantity of salts dissolved in a sample of water
consumer	an animal that consume biomass
Copepoda	a group of small, usually planktonic, crustaceans
CPOM	coarse particulate organic matter (organic fragments >1 mm in diameter)
Crustacea	the shrimp-like arthropods: crabs, prawns, amphipods, etc.
Cyanobacteria	(= blue-greens): a group of primitive unicellular or filamentous plant-like bacteria
Daphniidae	water fleas
Decapoda	the 10-legged crustaceans: crabs, lobsters, shrimps, prawns, etc.
decomposer	an organism that feeds on, or lives off, organic material, dead plants and animals
decomposition	the decay and breakdown of organic material, including the dead remains of plants and animals
density	(biological) a measure of the number of organisms per unit area
depauperate	poor, reduced
deposit feeder	an organism feeding on fine particulate organic matter that has been deposited on the substratum
detritivore	an animal that consumes detritus (q.v.)
detritus	dead and decomposing plants and animals
diatom	any of a group of unicellular or colonial algae with siliceous "shells" or frustules
diel	daily
Diptera	the two-winged insects, the flies and mosquitoes, larvae of many families are aquatic
diurnal	during the day
dystrophic	with insufficient nutrients
EC_{50}	(= median effective concentration): the concentration of a toxin at which a specified effect is observed in 50% of the population
ecology	the study of the interrelations between organisms and their environments
ecosystem 'health'	an estimate of the extent of ecosystem integrity
ecosystem	interacting organisms plus their environment
ecosystem functioning	the combined effect of the processes, such as photosynthesis, respiration and nutrient cycling, that make an ecosystem a functional whole
ecotone	the intergrading border between two different types of ecosystem
ecotoxicology	the study of the effects of toxins on organisms and ecosystems
ectoparasite	an external parasite
effluent	that which flows out (usually waste water)
electron transfer	the final process of respiration during which energy and CO_2 are released
Elmidae	the riffle beetle
embryogenesis	the development of an embryo

LC50	= median lethal concentration: the concentration of a toxin at which 50% of the test population dies
LD50	the dose of a toxin at which 50% of the test population dies
lentic	of waters, not flowing
limnology	originally the study of lakes; now the study of the physical, chemical and biological properties of inland waters
lipid	fat
lithosphere	the rocky crust of the Earth
logging	chopping down of trees
lotic	of waters, flowing
lower reaches	the downstream part of a river
LT_{50}	the time taken for half of the test population to die during tolerance testing
macroinvertebrate	an invertebrate (*q.v.*) visible to the naked eye
macrophyte	a large plant
macroscopic	visible to the naked eye
major ions	those ions (calcium, magnesium, sodium, potassium, bicarbonate, carbonate, chloride and sulphate) that usually *form the bulk of the total dissolved solids (q.v.) in inland* waters
maritime	pertaining to the sea
mayfly	ephemeropteran (*q.v.*)
median tolerance limit	the concentration of a toxin at which 50% of the test population dies
median effective time	the time taken for an observed effect to occur in half of a test population
median survival time	the time taken for half of the test population to die in tolerance testing
median lethal concentration	*see* LC_{50}
median lethal time	the time taken for 50% of a test population to die at a given concentration of toxin
mercurial	relating to mercury
mesic	moderate
mesocosm	an artificial experimental stream or lake, or an artificially enclosed part of a stream, lake or the sea
mesotrophic	with moderate levels of nutrients
metabolic rate	the rate at which chemical reactions take place in a living body
metal	an element that is a good conductor of electricity
metalloenzyme	an enzyme containing a metal atom as part of its structure
micro-nutrient	a plant nutrient, such as phosphorus or nitrogen, required in minute quantities
microbe	a micro-organism (e.g. a bacterium or a protozoan)
microcosm	an experimental system, usually laboratory-based, that mimics part of the natural environment
microfauna	minute animals invisible to the naked eye
microorganism	any organism so small as to be invisible to the naked eye
middle reaches	of a river, the intermediate or foothill zone
midge	any of the small flies of the order Diptera
mineralization	I) the release of inorganic ions from oxidized organic material ii) the increasing saltiness of soils and rivers as a result of irrigation (= *salinization*)
minimum acceptable tolerance concentration	of toxic substances, the theoretical concentration that falls between the highest concentration showing no effect and the next highest concentration showing a toxic effect, when compared to controls
mite, water	a small aquatic arachnid (*q.v.*)

mobilization	the process of becoming movable or soluble, and therefore readily available
Mollusca	the shell-bearing animals: snails, mussels, squid, etc.
mutagenic	causing mutations
natural environment	with regard to rivers, aquatic ecosystems and those ecosystems dependent on them
necrosis	the death of living tissue
nephelometry	the measurement of turbidity
nitrogen fixation	the process of converting gaseous N_2 to soluble forms (NO_2, NO_3) by bacteria
non-point	diffuse
NTU	nephelometric turbidity unit: the unit used for measuring turbidity
nucleic acid	a long-chain organic molecule found in the nucleus of a cell: DNA and RNA
nutrient	in aquatic biology an element whose scarcity can limit plant growth (e.g. compounds of nitrogen, phosphorus)
nymph	any of the immature stages of those insects (e.g. dragonflies) that do not have a pupa in the life cycle
Odonata	dragonflies and damselflies
Oligochaeta	earthworms
oligotrophic	low in nutrients
organic	of chemical substances, containing carbon
organism	a living thing
Oribatidae	a group of small, hard, seed-like mites with some aquatic members
orthophosphate	a form of soluble inorganic phosphate
osmotic balance	the balance between the amount of water and the concentration of solutes on either side of a membrane
overturn	of lakes, usually in autumn, the mixing of epilimnic and hypolimnic layers
oxic	with oxygen
P:R ratio	the ratio of production (photosynthesis) to respiration in an aquatic ecosystem
pan	a broad, shallow, sediment-filled basin that receives water in the rainy season
pathogen	a disease-producing organism
pathological	relating to disease
perennial	permanent: lasting throughout the year
periphyton	the "slime" of (usually) microscopic organisms growing on rocks, plants, etc., in the water
perturbation	disturbance
pest	a troublesome plant or animal
pesticides	substances that kill pest organisms
pH	the negative $\log_{10}$ of the hydrogen ion activity: a measure of acidity (pH <7) or alkalinity (pH >7)
photostable	not broken down by light
photosynthesis	the process whereby plants convert CO_2 and water into large, energy-rich, organic compounds
Physidae	a group of gastropod (*q.v.*) snails
physiology	the study of the functioning of living organisms
phytoplankton	plant plankton (*q.v.*) (includes many single-celled algae)
piscicide	a substance that kills fish
planarian	a turbellarian (*q.v.*) flatworm

plankton	the small organisms drifting or floating in water
Planorbidae	a group of gastropod snails
Platyhelminthes	the flatworms, tapeworms and flukes
Plecoptera	the stoneflies
pollutant	a substance that contaminates
pollution	defilement: unfavourable alteration of our surroundings, normally as a result of human actions; the presence of any foreign substance(s) that impair the usefulness of water
Polychaeta	bristle worms
polyphenols	large, complex, acidic organic molecules, characterised by a high proportion of phenol groups and often imparting a dark colour to water
precipitation	rainfall
predator	a carnivore that hunts its prey
primary producers	plants: the organisms that produce biomass, usually by photosynthesis
primary consumers	herbivores: animals that feed on plants
pristine	unaffected by human activities
productivity	the extent to which something (usually plant biomass) is produced
Protozoa	minute acellular organisms including flagellates, ciliates and amoebae
receiving water	a water body receiving an effluent
receiving water quality objectives	the water quality towards which a regulatory body will aim or strive
recruitment	the adding to a population of a new group of individuals, usually by reproduction
redox reaction	an oxidation/reduction reaction
reference conditions	the benchmark conditions of water quality and other environmental indicators against which the conditions at a particular site can be rated
Reserve	in South African water law, the amount and quality of water that must be kept aside for maintenance of an ecosystem
residue	remains; here referring to the breakdown products of organic molecules such as insecticides
respiration	I) *cellular* respiration: the metabolic process whereby energy-rich substrates are oxidized ii) "respiration": the exchange of gases (O_2 and CO_2) between an organism and the external medium
riffle	a shallow, fast-flowing reach of a river with turbulent flow and broken water
riparian	related to the river bank
Rotifera	a group of minute "wheel animals"
(natural) selection	the process whereby populations of organisms change over generations as a result of environmental pressures
salinity	"saltiness": the mass of dissolved inorganic solids in a kilogram of water (usually sea water)
salinization	the process whereby the saltiness of soils and rivers increases, usually as a result of human activities
Salmonidae	the group of fishes to which the trout and salmon belong
sandstone	a type of sedimentary rock
scraper	an animal that feeds by scraping fine material off the surfaces of rocks, plants, etc.
secondary consumer	an animal that feeds on herbivores
secondary plant compounds	complex organic substances forming 'humic substances' when leached into water

sediments	the soft sands, silts and muds on the bottoms of rivers and lakes
sessile	not movable: attached to the substratum and/or without a stalk
shale	a type of sedimentary rock
shredder	an invertebrate that feeds on small particles of organic matter chewed off large particles of detritus (usually leaves)
siltation	becoming silted up
Simuliidae	blackflies, larvae of which are common filter feeders in streams
spate	small flood
species, biological	a group of interbreeding individuals having common characteristics and being reproductively isolated from other groups
species richness	the number of species
species diversity	a measure of the number and relative abundance of species (see species richness)
species, chemical	the various chemical forms (e.g. differently charged ions) of an element
speciose	rich in species
stenothermal	able to withstand only a narrow range of temperatures
stoneflies	see *Plecoptera*
stratification	in limnology, the separation of the water in a lake into epilimnion and hypolimnion (*q.v.*)
stream order	an indication of the size of a river according to the number of tributaries (a stream with no tributaries is of stream order 1; two tributaries joining form a stream of order 2 ...)
substratum	the material that constitutes the bottom of a river or lake or the sea
suspensoids	particles suspended in the water column
swamp	a wetland that supports trees
synergist	one acting together with another (in toxicology, a substance that magnifies the toxic effect of another)
taxonomy	the naming and classification of living organisms
TDS	*see* total dissolved solids
teratogenic	causing birth defects
tertiary consumer	an animal that feeds on predators
thermocline	in a stratified lake, the interface between epilimnion and hypolimnion (*q.v.*)
Thiaridae	a group of gastropod snails
tillage	ploughing
total dissolved solids	the total mass of material dissolved in a sample of water
toxicant	a poisonous substance
toxicity	toxic quality, the poisonous-ness of a substance
toxin	poison
trace element	an element occurring at < 1000 ppm in the earth's crust
Trichoptera	caddisflies
tripton	suspended inanimate particles
trophic level	a division of the food chain defined by the method of obtaining food (e.g. primary producer, primary, secondary or tertiary consumer)
TSS	total suspended solids
Tubificidae	a group of Oligochaeta (*q.v.*) that is very tolerant of polluted conditions
turbidity	an expression of the optical property of water that causes light to be scattered and absorbed (murkiness)
turbulence	physical disturbance of water

vascular plants	"higher" plants with roots, stems, leaves, etc.
vector	the "carrier" of a parasite: usually a species that is itself infected by the parasite
Viviparidae	a group of gastropod snails
water quality	the value or usefulness of water, determined by the combined effects of its physical attributes and its chemical constituents, and varying from user to user
water quality standard	a rule authoritatively establishing, for regulatory purposes, the limit of some unnatural alteration in water quality that is permitted or accepted as being compatible with some particular intended use or uses of water
weir	a low dam
wetland	an ecosystem where the nature of the soils and of the biota is dominated by water
zooplankton	animal plankton (*q.v.*)

1770051317